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Regional Ecological Risk Assessment: Theory and Demonstration

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Environmental Sciences Division
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HUNSAKER, C. T., R. L. GRAHAM, G. W. SUTER II, R. V. O'NEILL, B. L. JACKSON, and L. W. BARNTHOUSE. 1989. Regional ecological risk assessment: theory and demonstration. ORNL/TM-11128. Oak Ridge National Laboratory, Oak Ridge, Tennessee. 115 pp.

Society needs a quantitative and systematic way to estimate and compare the impacts of environmental problems that affect large geographic areas. This report presents an approach for regional ecological risk assessment that combines regional assessment methods and landscape ecology theory with an existing framework for ecological risk assessment. Risk assessment evaluates the effects of an environmental change on a valued natural resource and interprets the significance of those effects in light of the uncertainties identified in each component of the assessment process. The components of regional risk are defined, and the similarities and differences between regional and local risk assessment are discussed in Chapter 1 of this report. Unique and important issues for regional risk assessment are emphasized; these include the definition of the disturbance scenario, the assessment boundary definition, and the spatial heterogeneity of the landscape. In Chapter 2 we present an in-depth discussion of possible endpoints for regional assessments and criteria for judging endpoints. The nature of the assessment problem influences the appropriate choice of endpoints.

A demonstration of a regional risk assessment is used to illustrate the components of the assessment framework, to test the utility of the approach, and to highlight unique aspects of regional assessment such as spatial heterogeneity, landscape pattern, and the need to link ecological systems through the use of models (Chapter 3). The environmental

disturbance assessed is the impact of ozone-induced pest infestations on land cover, wildlife habitat, and water quality in the Adirondacks of New York. The Adirondack region was selected for the demonstration because of the availability of data and a suitable lake water quality model. A method for incorporating probabilistic response into spatial, regional models is illustrated and tested. Results indicate that consideration of landscape pattern is necessary for regional ecological risk assessment.

1. AN INTRODUCTION TO REGIONAL ECOLOGICAL RISK ASSESSMENT

1.1 INTRODUCTION

The objective of risk-based ecological assessment is to provide (1) a quantitative basis for balancing and comparing risks associated with environmental problems and (2) a systematic means of improving the estimation and understanding of those risks. In ecological risk assessment, uncertainties concerning potential environmental effects are explicitly recognized and, if possible, quantified. A better understanding of risks associated with an environmental problem is achieved by comparing the magnitudes of uncertainties in different steps of the causal chain that links the initial event (e.g., release of a toxic chemical) and its ultimate consequence (e.g., alteration of an ecosystem). Ecological processes operate at a variety of scales in space and time. Many environmental problems impact large geographic areas (e.g., acid deposition, non-point-source pollution, and increased global CO_2), yet traditional concepts and methods in ecology and risk assessment are relevant mainly to single sites or small geographic areas. Effective long-term management and protection of valuable natural resources require a better understanding of how the scale of the environmental problem affects ecological processes and over what scales the effects should be monitored and examined.

Any risk assessment should be properly scaled for the environmental problem being analyzed. Problems and their associated risk assessments will exist along a continuum of spatial scales; but for ease of discussion, we will divide that continuum into two classes--local and regional. Our differentiation is best illustrated by example. Local problems amenable to local risk assessments include (1) the effects of a single industrial effluent on water quality in a 1-mile stretch of stream and (2) the effects of harvesting practices on the habitat of an endangered species in a tract of a national forest. Regional counterparts to these local problems would be (1) the impacts on water quality in a river basin that will result from proposed industrial and municipal discharges and projected land use in the next 10 years and (2) the effect of forest management practices on the survival of the spotted owl in the entire Pacific Northwest. In these latter two examples, both the cause and the consequence of the environmental problem are regional, however, a number of factors may cause an environmental problem to become regional. Regional risk problems can also be caused by local phenomena that have a regional consequence (e.g., single-source pollutants that become widely dispersed, such as the radioactivity from Chernobyl). The desire to protect a population, species, or ecosystem type that is widely dispersed could be the reason for performing a regional risk assessment. In this case, multiple local factors create a regional problem.

We will first describe a two-phase approach for doing regional ecological risk assessment and discuss the key components of the first phase of that approach. We will then discuss sources of uncertainty in all phases of regional ecological risk assessment.

1.2 REGIONAL RISK ASSESSMENT APPROACH

Our approach to regional risk assessment is derived from the scheme for ecological risk assessment described by Barnthouse and Suter (1986). The key components of risk assessment include (1) selection of endpoints, (2) qualitative and quantitative description of the sources of the disturbance (e.g., locations and emission levels for pollutant sources), (3) identification and description of the reference environment within which effects are expected, (4) estimation of spatiotemporal patterns of exposure using appropriate environmental transport models, and (5) quantification of the relationship between exposure in the modified environment (reference environment) and effects on biota. Finally, all of the previous steps are combined to produce the final risk assessment. The risk assessment expresses the ultimate effects of the source on the endpoints in the reference environment and interprets their significance in light of the uncertainties identified in each component of the assessment process.

To express some of the concepts pertinent to regional risk assessment, we found it necessary to modify some conventional risk assessment terms developed for assessment of chemicals (explained in Table 1-1). We expand the term <u>exposure</u> to include both the chemical and physical (e.g., loss of habitat) exposures a target organism might experience in the modified environment. We also introduce the term <u>disturbance</u> to describe the phenomenon that creates risk--that is, the pollutant or activity (source) potentially subjected to legal regulation--and the disruptive influence it has on the system.

Table 1-1. Regional risk assessment terms

Term	Definition	Example
Disturbance	Pollutant or activity and its disruptive influence on the ecosystem containing the endpoint[a]	Forest cutting practices that eliminate critical habitat for an endangered species
Endpoint	Environmental entity of concern and the descriptor or quality of the entity	Extinction of an endangered species
Source terms	Qualitative and quantitative descriptions of the source of the disturbance	Forest cutting practices and the laws and economic factors that influence them in the Piedmont
Reference environment	Geographic location and temporal period	Piedmont of the United States in the next ten years
Exposure	Intensity of chemical and physical exposures of an endpoint to a disturbance	Amount of habitat, for an endangered species, that is lost

[a]Equivalent to hazard in toxicological assessment.

Both regional and local risk assessments can be thought of as having two distinct phases--the definition phase and the solution phase of the problem (Figure 1-1). Significant differences exist between local and regional risk assessments in both these phases. In the definition phase, the endpoint, source terms, and reference environment are defined. The first step in local risk assessment is usually the selection of endpoints; then the source terms are defined, and the reference environment is described. This sequence is fairly distinct. In regional risk assessment, however, the initial concept of the problem or disturbance usually is more nebulous, and the interactions between the components of the definition phase are often more complex. Understanding of the disturbance must be refined, taking into account not only the ecological processes of interest but also the social, economic, and institutional processes significant to the disturbance. Then endpoints are selected, source terms are developed, and the reference environment (in this case the region) is described. The definition of these three elements is an iterative, rather than sequential, process.

In the solution phase, exposure and effect are assessed and then combined to determine probability of risk. In this phase, regional assessments differ from local ones primarily in two ways. First, the models used in the exposure and effects assessment must be regional; local models may have to be adapted to larger geographic regions (Dailey and Olson 1987). Second, the exposure or effects assessment must account for uncertainty that may arise because of spatial heterogeneity, a feature that is not significant in local assessments.

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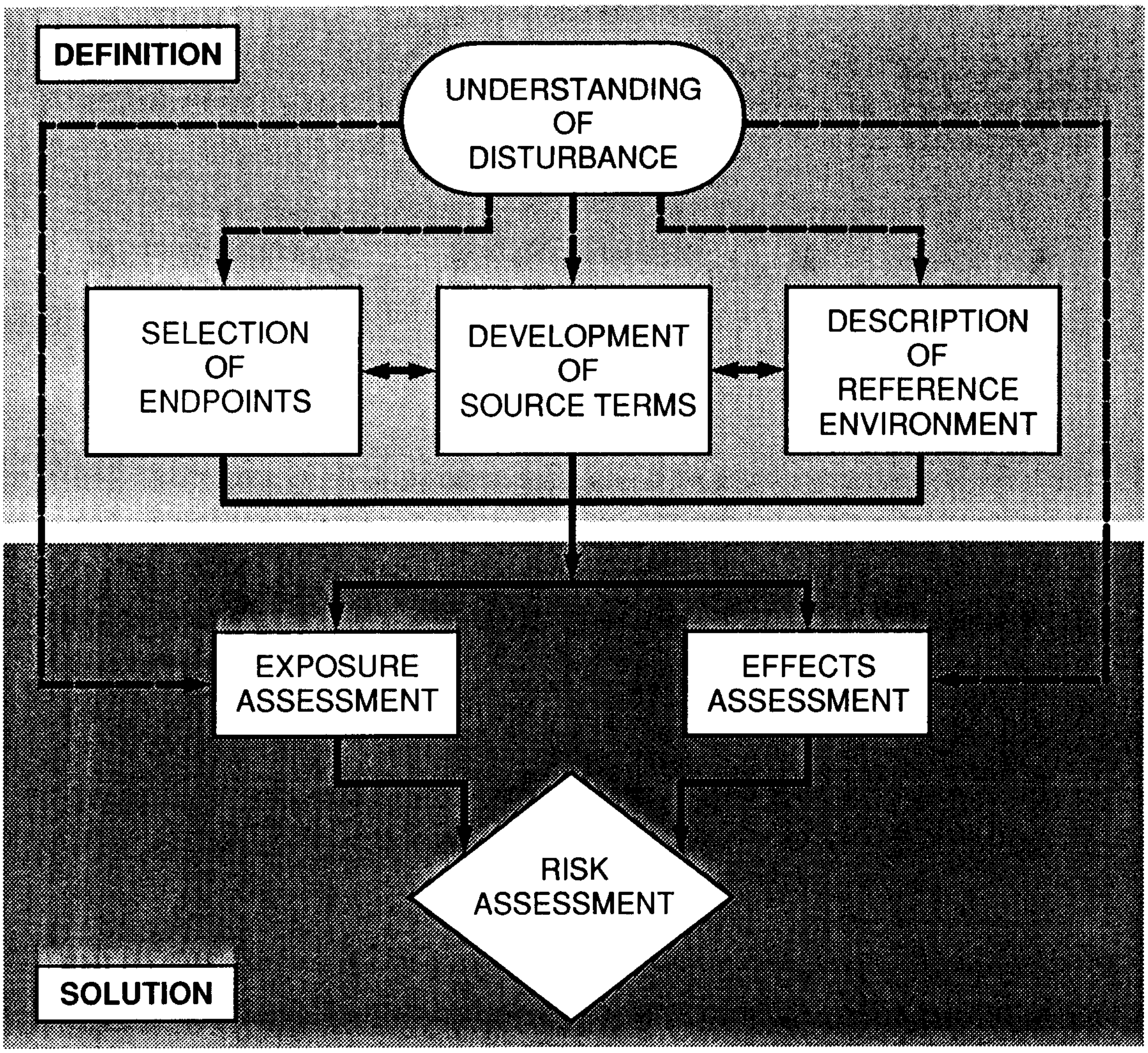

Fig. 1-1. The two phases of regional risk assessment: the scoping or setting up of the problem and the solving of the problem.

We foresee that the definition of the regional ecological problem will be iterative because the source terms, endpoint, and reference environment are all interdependent. Any refinement in one of these components will affect the others. Moreover, with increased understanding of the interactions between these components, there will be less uncertainty in the assessment. Because the links between these elements will be unique to each specific regional assessment, no single paradigm will be satisfactory for all environmental issues. The degree to which the setup of the risk analysis problem is legislatively defined will very much influence both the cost and technical difficulty of the assessment and its final outcome.

1.2.1 Description of Region

For a regional assessment of an environmental problem to be effective, the spatial and temporal boundaries of the reference environment must be defined appropriately for the disturbance and the endpoint. A region is a spatially extended, nonhomogeneous geographic area; it is nonhomogeneous in the sense that smaller spatial units exist within the region that are more homogeneous than the region. For ecological assessments the functionally defined region is the most useful. A functionally defined region in an ecological sense is one in which the boundary is determined by physical or biological ecosystem processes important to the disturbance being evaluated (e.g., watersheds, airsheds, or physiographic provinces). Often, assessment boundaries are determined by nonecological factors such as political boundaries, available data, the influence of interest groups, or the estimated concentration of a pollutant. A region can be defined by the area in which the concentration of a pollutant exceeds some standard. An assessment boundary could also be a hybrid of the functionally defined region and another factor that influences the boundary. When an assessment uses a region that is not functionally defined, the results will likely be useful only for the problem addressed and cannot be extrapolated to other regions.

The geographic area in which the endpoint experiences the disturbance may be a subset of the geographical area that encompasses the disturbance (Bormann 1987). The latter possibility may occur when economic/political processes have an important affect on the disturbance, as in the case of atmospheric deposition. Because air masses cross political boundaries, risk assessment aimed at protecting the forests of Germany cannot be conducted without considering the emissions of neighboring countries. In some ways a functionally defined region is a platonic ideal as our understanding of any disturbance will be fraught with uncertainties both conceptually and analytically. In reality, the appropriate region for a problem may not become apparent until part or all of an assessment is complete. To some extent, choice of boundaries will ultimately be limited by current knowledge and funding, even if the intent is to define a functional region. More research is needed on how to integrate ecological, social, and economic data in order to determine the boundaries of assessment regions.

1.2.2 Selection of Endpoints

For any risk assessment, endpoints must have biological relevance, be important to society, have an unambiguous operational definition, and be accessible to prediction and measurement (Barnthouse and Suter 1984). Definition of the endpoint includes the environmental entity and the descriptor or quality of the entity. The descriptor may reflect the socially/politically unacceptable level of effect of the disturbance on the entity. An entity can be an organism or a medium such as air, water, or soil. An endpoint can be both legislatively (e.g., criteria or standard) and functionally defined. Endpoints for regional risk assessment are briefly discussed here; a detailed discussion of endpoints is contained in Chapter 2.

Endpoints for regional risk must be regional in scope. Regional risk endpoints can be hazard/exposure oriented or effects oriented. Exposure-oriented endpoints are such things as chemical concentrations in media or biota. Effects-oriented endpoints include loss of a population or changes in system properties such as productivity or albedo.

Endpoints must be defined in terms of observations that can be made over large areas and long time periods. Thus, monitoring indicator species or critical habitat is a useful concept and an economical intermediate for regional endpoints. For terrestrial systems possible endpoints include percent cover of different vegetation types, forest productivity or composition, or presence of a species. Vegetation can be measured by remote sensing or periodic surveys of vegetation and land use.

Observations for regional aquatic assessments might consist of data on water quality and species composition, often spaced at relatively long temporal intervals (with respect to life cycles for aquatic biota). Endpoints might be expressed as presence or absence of particular species of interest, or in terms of system-level indicators. Examples include the frequency of lakes or nth-order streams in which brook trout (or some other important species) become extinct, percent areal reduction of Spartina in salt marshes, and percent of nth-order streams dominated by pollution-tolerant organisms.

As noted before, we expect that emergent properties exist at the regional or landscape scales (Allen, O'Neill, and Hoekstra 1984). Indices of landscape pattern that could be measured at the scale of a landscape and that are reflective of important ecosystem concepts or processes relevant to the disturbance may prove to be useful regional endpoints. Example indices include dominance, contagion (degree to which the landscape is dissected into small patches or aggregated into large, continuous patches), fractal dimension (index of complexity of shapes on the landscape), and amount of edge (Krummel et al. 1986, O'Neill et al. 1988). Because such indices might be calculable from remotely sensed imagery, they might also be useful in long-term monitoring of processes.

1.2.3 Development of Source Terms

Developing source terms may be more difficult for regional problems because they often involve multiple sources that vary in both space and time. Source terms are the qualitative and quantitative descriptions of the sources of disturbance (e.g., locations and emission levels for pollutant sources). As noted before, source terms include social, institutional, and economic factors that affect the pollutant or activity. Compared with the problems previously addressed in environmental risk assessments (effects of one or a few risk sources on local populations and ecosystems), regional environmental problems involve risk sources that affect large areas, usually over long periods of time. Sometimes regional problems can have acute effects also (e.g., acute lethality to fish due to a basin-wide combination of reduced runoff, increased water withdrawal, and overloaded sewage treatment plants). Regional assessments of effects of air pollution (including acid deposition) on terrestrial and aquatic systems involve multiple pollutant sources that affect thousands of square kilometers. Concentrations and deposition rates have short-term cycles, but effects become observable only after years of exposure. Basin-level water quality problems are similar.

1.3 SOURCES OF UNCERTAINTY IN REGIONAL ASSESSMENTS

Regional ecological risk assessment involves, for the most part, the same set of sources for uncertainties involved with local risk assessment. The relative importance of a given uncertainty depends on the disturbance and the endpoint. Some components of uncertainty are relevant to both the definition and the solution phases of a regional risk assessment, whereas others are important to only one phase. Uncertainties related to source terms and boundary definitions are relevant to the problem definition phase, whereas uncertainties related to model structure and model parameters are relevant to the solution phase. Uncertainties related to temporal scale and spatial heterogeneity are important to both phases of the assessment. All the uncertainties are combined in the final risk assessment.

The quantification of uncertainty for local ecological assessments has only recently received serious attention (Suter et al. 1987), and quantification of uncertainty for regional assessments is just developing (Kamari et al. 1986, Cosby et al. 1987). Uncertainties may remain quite large in regional assessments, and there may be no practical way to reduce that uncertainty regardless of cost. Risk assessments centering on disturbances that are highly dependent on economic, social, and/or political factors are likely to fall into this category. If regional risk assessments are to be economical and useful, recognition of the importance of these factors early in the problem definition is critical.

1.3.1 Scenario Definition

Sometimes it is difficult to define source terms for a disturbance, especially in predictions for the distant future. When some component of the disturbance is highly uncertain scenarios are a tool for bracketing the potential range of the disturbance or some component of it. Typically, several possible sets of scenarios--that is, source terms, reference environments, and endpoints--are considered. Scenarios will likely be used in regional risk assessments when considerable uncertainty exists about the disturbance (e.g., climate change or future mix of energy production). The results of such risk assessments are conditional on the events in the scenario. Thus it is important to try to select scenarios that take into account probable events. For regional studies, the absolute uncertainty predicted for a given scenario might not be very useful, but the comparisons between the relative uncertainty from the analysis of each scenario will be useful to the decision maker.

1.3.2 Boundary Definition

The least amount of uncertainty occurs when the "true" geographic boundary for the disturbance is known (Allen et al. 1984), as with a pollutant whose transport and fate are well defined. Boundary definitions become a problem when the functional region crosses political boundaries. Once a boundary is set and analysis proceeds, the ability to assess the uncertainty introduced by the choice of the boundary is lost. Boundary problems could especially add to the uncertainty of an assessment if there is an omission of an important source, a component of an endpoint, or a process that influences the relationship between a source and an endpoint. For some problems, the error associated with the definition of the spatial and temporal boundaries for a region should be evaluated by estimating the risks under several different boundary definitions.

1.3.3 Data and Model Availability

The availability of data bases and models is a critical factor in the quality of an assessment. Although uncertainties in models and data arise in local risk assessments, they may become more critical in regional ones. Regional studies can be classified into several types: classification or inventory studies, facility siting studies, NEPA documents, and more recently, studies that are similar to a regional risk assessment (Hunsaker et al. 1986, Hunsaker et al. 1987, Kamari et al. 1986, Cosby et al. 1987). The ability of a model to represent environmental processes at the spatial and temporal scales of interest is a fundamental issue. Few regional-scale biological models exist. In most instances, either local models will have to be adapted to larger regions (Solomon 1986, Dale and Gardner 1987, Thornton et al. 1987, Cosby et al. 1987), or entirely new models will have to be developed (Emanuel et al. 1985a, Emanuel et al. 1985b, Hunsaker et al. 1986).

Few regional-scale biological models exist. In most instances, either local models will have to be scaled up or entirely new models will have to be built. Both approaches have been used in modeling the effect of elevated CO_2 on regional vegetation and in assessing the impact of acid deposition on lake chemistry. Using the former approach, a stand-level simulation model was applied at numerous locations within a region to explore the transient response of regional forests to a new climate (Solomon 1986). This approach is also being investigated for modeling lake acidification (Thornton et al. 1987, Cosby et al. 1987, Dailey and Olson 1987). Using the latter approach, an empirical relationship was developed between biome vegetation and climate. This relationship was then used to project a new pattern of global vegetation under a new global climate (Emanuel et al. 1985a, Emanuel et al. 1985b). In a similar vein, Hunsaker et al. (1986) developed an empirical model that predicted lake acidity from watershed characteristics for the Adirondacks.

Some regional-scale models may well be impossible to validate in the traditional sense. In such cases, quantification of the error associated with the model's structure will be difficult. Examples of such models include those that predict a modified environment as a result of events that have never occurred, such as a major transportation accident involving nerve gas, an extreme climate change, or any situation in the distant future. Such models can only be evaluated for reliability and appropriateness (i.e., verified). Klemes (1985) discusses model transferability and presents a hierarchical scheme for systematic testing of models. Sometimes it is useful to compare models that purport to predict the same condition or effect (Thornton et al. 1987, Turner 1987b); if the models give similar results, then confidence in their prediction is improved. But sometimes only one model is available. Another validation technique is to use, as the evaluation data set, the portions of a known data distribution that are representative of the conditions that the model needs to predict. For example, a model designed to predict the effects of climate change might be validated using data on the effects of observed climatic extremes--the wettest/driest and warmest/coolest portions of meteorological records.

Uncertainties associated with parameter values can be partially resolved through standard uncertainty test procedures (Gardner 1984, Hoffman and Gardner 1983). Parameter uncertainty includes both natural variability and uncertainty due to lack of knowledge. In regional ecological risk assessments, uncertainties that arise from the inherent variability and heterogeneity of natural populations and ecosystems are especially important. Population and ecosystem data contain inherent variability that no amount of monitoring will reduce completely.

The quality, acquisition, and use of data can dramatically affect the cost of an assessment and can contribute to uncertainty. Historically, a major portion of the time spent on a regional assessment was devoted to locating the data and building an integrated data base. Olson (1984) prepared a review of environmental and natural resource data bases and identified 24 federal agency clearinghouse, referral, or data centers that maintain inventories of machine-readable data files. He concluded that ecological and biological data appear to be more widely dispersed and less standardized than hydrologic, climatic, or other abiotic data. Large, integrated data base systems that provide selected data stored in compatible spatial and temporal formats, with associated analysis and mapping capabilities to conduct integrated studies, are less common. Such systems include the Department of Energy's GEOECOLOGY and SEEDIS, the Council of Environmental Quality's UPGRADE, the Environmental Protection Agency's GEMS and ADDNET, and DATAGRAF (Merrill 1982, Olson et al. 1987). Long-term maintenance of integrated data bases is difficult because of funding cycles and perceived need. Therefore, for some problems a significant amount of effort may have to be expended on data integration and quality assurance. The paucity of adequate data sets both spatially and temporally has limited our ability to evaluate landscape ecology theory.

Remote-sensing technology offers a unique, synoptic view of a region. Unfortunately, with the exception of weather forecasting, it is still not a routine process to convert remotely sensed spectral data into useful, biologically interpretable data which can be merged with other environmental data. Nonetheless, the technology has been successfully applied to environmental issues such as assessing forest damage (Rock et al. 1986), monitoring shifts in marine productivity (Perry 1986), mapping vegetation cover (Tucker et al. 1986), and monitoring drought (NASA 1987). The use of remotely sensed data in environmental applications has been hampered by the lack of biologists trained to use this type of data. With the advent of sensors expressly designed for vegetation analysis, this situation is changing. Remote sensing will become an increasingly important tool as we move into regional and global studies (Greegor 1986).

Data manipulation and extrapolation can also contribute to uncertainty because error may arise during the process of sampling at a particular spatial and temporal frequency (grain and extent), extrapolating from point data to contour data, and aggregating and disaggregating data. Point data for large geographical regions are often uneven in quality and distribution. For example, one state may gather water quality data with one technique, and another state may use another technique or a different sampling frequency. The classification of geographic areas according to the relative homogeneity of one or more environmental attributes can be extremely useful in reducing uncertainty if the classification scale is appropriate to the disturbance. Ecoregions are examples of geographic classifications (Bailey 1983, 1987; Omernik 1987; Rohm et al. 1987). However, the contribution to assessment uncertainty from such classification needs further investigation because classification or aggregation of data could mask spatial heterogeneity that is significant to a realistic evaluation of the problem (McDaniel et al. 1987).

1.3.4 Temporal Dynamics and Scale

Uncertainty will increase if the risk assessment does not encompass disturbance dynamics at the appropriate temporal scale. If exposure has considerable temporal variation within a year, mean annual values of exposure or monthly averages may not reflect the impact on the endpoint. For example, episodic events of low pH associated with snow melt are of very short duration but can nevertheless determine trout survival. In this case, the extremes for pH and aluminum, not the means, are of critical importance, and the use of monthly averages would result in a highly uncertain or even meaningless estimate of effects on fish. Instead, hourly measurements for aquatic systems are needed. The appropriate temporal scale may vary with different aspects of the same disturbance. In the preceding trout example, knowing sulfur deposition on a daily, rather than on a monthly, basis would probably not reduce the uncertainty in the risk assessment.

1.3.5 Spatial Heterogeneity

Spatial heterogeneity can be a major source of uncertainty in regional ecological risk assessment. Most ecological modeling has not included spatial relationships, and there are no accepted measures of landscape pattern or heterogeneity that can be linked to processes occurring at a landscape scale (Bormann 1987). Although spatial heterogeneity is not necessarily a factor in all regional risk assessments, it can contribute to uncertainty in some situations. Thus, one must first ascertain if spatial heterogeneity is likely to influence the projected outcome of the disturbance. If it is, then spatial heterogeneity must be accounted for in the assessment.

Some disturbances can be viewed as an aggregation of local disturbances. In such cases, the regional risk is simply the sum of the local risks. For instance, estimation of the number of acidic lakes in the United States has been treated as an aggregate problem. Therefore, the United States was stratified into (1) regions; (2) homogeneous subregions with respect to physiography, vegetation, climate, and soils; and (3) alkalinity classes. Then, a statistical sample of all the lakes in a stratum was used to predict a regional and, eventually, a national value for the number of acidic lakes (Lindhurst et al. 1986). If, however, one or more properties associated with the disturbance become apparent only on a regional scale, then treating the disturbance as an aggregation of local effects is inappropriate. The impact of sewage on water quality, for example, is a function of not only the amount of sewage but also the quality of the water upstream of the discharge. Thus, when the connectedness of the hydrologic system is an important feature of the disturbance simply summing local risks is not an adequate assessment.

Aspects of spatial heterogeneity that might influence ecological risk include patch and population sizes, ratio of patch edge to interior, distance between patches, and appropriate spatial resolution. Because a better understanding of these aspects is essential for regional ecological assessment, they are discussed separately at greater length.

The size distribution of habitat patches or populations in a region may affect the impact of a disturbance (Turner 1987a, Sharpe et al. 1987, Hayes et al. 1987). For example, forest bird species richness in a temperate agricultural landscape is a linear function of the log of the size of remnant forest patches (Freemark and Merriam 1986). Thus a disturbance which reduced forest patch size would effect species richness differently for different size patches. All species require habitat of some minimal area; certain populations are likely to disappear if that area becomes too small (Noss 1983, van Dorp and Opdam 1987). Furthermore, some ecosystem functions (e.g., wetland ability to remove pollutants) may disappear when the system is reduced beyond a certain point. Ignoring the size distribution of patches or populations may increase the uncertainty associated with the risk assessment when this type of spatial heterogeneity is important.

The ratio of edge to interior of landscape elements such as lakes and forests may be important in assessing the ecological risk of some disturbances. For example, the ratio of forest edge to interior has a profound effect on the magnitude of blowdown experienced in the Pacific Northwest (Franklin and Forman 1987). Cutting patterns that increase that ratio will increase blowdown even though the total area of cut forest may remain the same.

The distance between similar units of land or phenomena may also affect the outcome of a regional ecological risk assessment. For instance, distance between similar habitats may affect the ability of a species to migrate, which, in turn, may affect its ability to maintain a stable regional population under a given level of disturbance.

For each disturbance there is a particular spatial scale at which uncertainty is minimized or the disturbance is most clearly seen (Allen et al. 1984). Landscapes can be compared to pointillistic paintings. If the viewer is too close (at too fine a resolution), the objects of interest cannot be seen. If the viewer is too far away (at too coarse a resolution), again, the objects of interest cannot be seen. It will be important in regional risk assessment both to identify the optimal spatial scale for viewing and collecting data and also to understand how the scale at which the landscape is viewed alters uncertainty.

1.4 CONCLUSION

Although regional studies have been performed for many years (McHarg 1969, Levenson and Stearns 1980, USDOE 1981, Klopatek et al. 1981, Westman 1985), the ecosystem properties that are important for regional scales are still poorly understood (Meentemeyer and Box 1987). The degree to which these properties are significant in regional risk assessment is even less understood. To define the uncertainty associated with ecological risk assessments, we need to consider, as well as possible, the implications of scale to the risk assessment.

Regional risk assessment has some attributes in common with local risk assessment but has others that are unique. The general theoretical framework for doing the two types of environmental risk assessment is the same. Both have two phases: first, the problem definition, in which the endpoint, source terms, and reference environments are defined and described; and second, the problem solution, in which the exposure and effect on the endpoint are assessed by using models and the risk and its associated uncertainty are determined. Regional risk assessment differs in (1) the extent of interaction between the source terms, endpoints, and reference environment and (2) the degree to which boundary definition and spatial heterogeneity are significant in determining uncertainty. Although local risk assessments involve the development of data bases and the use of models, these steps may be more significant in regional risk assessments. Few regional-level data bases of biological variables exist; furthermore, unique problems arise in aggregating or integrating dissimilar local data into regional data bases. Regional models are much less common and are potentially often likely to be difficult to validate.

Although most of the fundamentals are in place for doing regional risk assessment, research is still needed on both theoretical and applied issues. Little is known about the influence that data aggregation has on uncertainty in model parameters. Questions about this influence invariably arise in regional studies with large data bases. Ecological hierarchy theory, ecoregion definitions, and multivariate and spatial statistical techniques will be useful in assessing the significance of data aggregation. Research on the appropriate models for regional studies needs to continue. We need to know under what circumstances it is appropriate to adapt a local model to a region and how to do so. Our tools for describing landscape pattern are still experimental. The development of landscape pattern indices that capture important ecological processes at the landscape scale could significantly simplify regional monitoring. However, this development will require a more complete understanding of the interaction between landscape pattern and ecological processes. Some of the more recent technological tools--such as geographic information systems, satellite sensors that capture biologically significant spectral patterns, and super computers that can process large spatial arrays--will be useful for addressing the theoretical and applied research issues that the regional scale poses. The simple lack of adequate spatial and temporal data for large geographic areas severely limits regional risk assessments. Many tools and ideas exist, but they need to be tested and refined before regional ecological risk assessments can become an effective tool for managing and protecting natural resources.

2. ENDPOINTS FOR REGIONAL ECOLOGICAL RISK ASSESSMENT

2.1 INTRODUCTION

Regional ecological risk assessment is a new activity concerned with describing and estimating risks to environmental resources at the regional scale or risks resulting from regional-scale pollution and physical disturbance. Examples include acid rain effects, Antarctic ozone depletion, and pollution of a river basin by multiple point and nonpoint pollution sources. Because of the apparent increase in the number of regional problems and the recognition of the value of a regional perspective in environmental regulation, the recognized need for regional risk assessment is increasing. If regional assessments are to be performed efficiently and effectively, it is necessary to consider how each of the components of a risk assessment must be adapted to address regional-scale problems. The component addressed in this paper is the endpoints, those characteristics of valued environmental entities that are believed to be at risk.

Ecological risk assessments begin with three activities that define the nature of the problem to be assessed: choosing endpoints, describing the environment, and describing the hazard. These are followed by a formal analysis of the problem consisting of exposure assessment, effects assessment, and integration of the exposure and effects assessments to estimate the probability and level of effects. In a process called risk management, the results of the risk assessment are considered along with economic, technological, and political considerations to arrive at a decision. Each of these component processes should be coordinated. This paper describes two different expressions of endpoints, presents criteria for judging endpoints, presents sets of endpoints that are potentially useful in regional risk assessments, judges them by the criteria, and discusses how the nature of the assessment problem affects endpoint choice.

2.2 TYPES OF ENDPOINTS

Some confusion has occurred in environmental risk assessment because the term endpoint has been used to describe two related but distinct concepts. To avoid this confusion we have distinguished assessment endpoints from measurement endpoints. Assessment endpoints are a formal expression of the actual environmental value that is to be protected. The output of a risk assessment is an estimated probability of occurrence of a dichotomous assessment endpoint (e.g., probability of extinction of a species) or an estimated relationship between probability and magnitude of a scalar assessment endpoint (e.g., probability that the number of fishless lakes will be greater than X). These expressions of effects on assessment endpoints are the input to the risk management process. Assessment endpoints must be valued by society, but they are not ultimate values. Rather, they are the highest values that can be formally assessed. In regional risk assessment, the ultimate value is the quality of life provided to the region's inhabitants, which is an indefinably function of the region's ability to provide food, clean water and air, aesthetic experience, recreation, and other services without floods, property-damaging fires, and other disservices. Such ultimate values fall in the domain of risk management where risk assessment results are considered along with political, economic, and ethical values.

A measurement endpoint is an expression of an observed or measured response to the hazard; it is the empirical expression of the assessment endpoint. Measurement endpoints are typically simple statistical or arithmetic summaries of the measurement results. Examples are the median lethal concentration (LC_{50}), a point on a regression line fitted to concentration-response data, and relative abundance measures such as area of wetland per unit length of coast (WRI/IIED 1986). The term "test endpoint" is used in environmental toxicology, and measurement endpoint is simply an expansion of this concept to include expressions of field monitoring studies. In some cases, the measurement endpoint may be the same as the assessment endpoint. For example, if the endpoints for sugar maple decline are increased mortality and decreased sugar production, then sugar maple mortality rates and sugar production can be directly monitored and related to environmental conditions. Because the assessment endpoint may not be observable or measurable or because available or standard data must be used in an assessment, measurement endpoints are often surrogates for the assessment endpoints. For example, if the assessment endpoints are reductions in populations of largemouth bass and channel catfish and the hazard is an effluent containing aniline dyes, then a measurement endpoint might be an aniline LC_{50} for fathead minnows.

Although all risk assessments must have assessment endpoints, there may be no measurement endpoints. The assessment may be based on theory or assumptions about the relationship between the hazard and the assessment endpoints. For example, Krummel et al. (1984a) assessed the sensitivity of plant communities in western Kentucky on the basis of the distribution of SO_2 concentrations relative to the distribution of plant communities. Because they did not have the opportunity to measure SO_2 effects in the various communities and did not feel that the existing phytotoxicity data was adequate, the authors hypothesized two possible threshold concentrations for SO_2 effects and assumed that all communities were equally sensitive. The uncertainty introduced by the absence of effects measurements limited the assessment to suggesting areas that were worthy of study rather than actually predicting effects. In other cases, measurements may be unnecessary or impossible. For example, if the assessment endpoint for an assessment of a proposed power plant is the probability of exceeding an air quality standard, then there is no environmental response to measure and, assuming that good local meteorological data and source terms are available, models based on atmospheric theory are adequate predictors.

Unfortunately, in many monitoring programs, clear measurement endpoints are applied to vague assessment endpoints such as "are the things that we are measuring changing?" or "are the things that we are measuring different at these two sites?" Without a clear definition of why measurements are being taken, time and effort are wasted. If one monitors any aspect of the environment long enough, change will be seen, and if any two sites are sampled intensively enough, they will be found to be different. A clearly defined assessment endpoint not only indicates what is worth measuring but also how intensively it must be measured.

2.3 CRITERIA FOR ENDPOINTS

2.3.1 Assessment Endpoints

Criteria for good ecological risk assessment endpoints are listed in Table 2-1. First, an assessment endpoint should have societal relevance; that is, it should be an environmental characteristic that is understood and valued by the public and by decision makers. An equivalent term of art used in the EPA is "regulatory impact." In local risk assessments the most appropriate endpoints often are effects on valued populations such as crops, trees, game fish, birds, or mammals and these are likely to be important in regional assessments, also. Societal value is emphasized because assessments of risks to nematodes or aphids are unlikely to influence decisions. This is not to say that species and other environmental attributes that are not publicly valued or understood have no place in environmental risk assessment. Rather, if species that are not socially valued are particularly susceptible, then they must be explicitly linked to valued species or other valued environmental attributes. Societal significance is also diminished by the use of indices that integrate a composite of entities. For example, diversity indices combine species number and evenness in a biotic community into a single number. Typically such indices are not interpretable by decision makers or the public, hide useful information such as what species have been lost, and may be misleading.

It is desirable that the assessment have biological relevance. The biological significance of an effect is a function of its implications for

Table 2-1. Characteristics of good assessment endpoints

1.	Social relevance
2.	Biological relevance
3.	Unambiguous operational definition
4.	Accessible to prediction and measurement
5.	Susceptible to the hazard

the next higher level of biological organization. For example, the significance of infertility of individuals is determined by the resulting population reduction, and the significance of the loss of a major grazing species is determined by the ability of other grazers to functionally substitute for the lost species thereby sustaining the community structure. As a further example, physiological stress markers may indicate pollution exposure, but they are also a part of adaptation to varying environmental conditions and may have no long term implications for either organisms or populations. Biological significance may not correspond to societal significance. The abundance of bald eagles has clear societal significance but the near extinction of bald eagles in the contiguous United States apparently had no significance for the rest of the biota.

Assessment endpoints should have unambiguous operational definitions. Although phrases like "ecosystem integrity" and "balanced indigenous populations" adequately express the longing of legislators for a good natural environment, they are not suitable subjects for risk assessment. Exactly how do we know when an ecosystem has lost its integrity and what do we balance a population against? A complete operational definition of an assessment endpoint requires a subject (bald eagles or endangered species in general) and a characteristic of the subject (local extinction or a percentage reduction in range).

Assessment endpoints should be accessible to prediction and measurement. Prediction requires toxicity tests and statistical models for summarization and extrapolation of test results, measurements of responses of similar systems to similar hazards, or mathematical models of the response of the system to the hazard. An endpoint that can not be tested, measured, or modeled can not be assessed except by expert judgement (a notoriously weak foundation for risk assessment -- Fischoff et al. 1981). For example, sharks are not used in toxicity tests and good fisheries data for sharks are not available, so effects of pollution on sharks are not good assessment endpoints.

Finally, the assessment endpoints must be susceptible to the hazard being assessed. Susceptibility results from a potential for exposure and responsiveness of the organisms or ecosystem attribute to the exposure. In some cases, susceptibility will be known in advance because observed effects prompted the assessment. In other cases, where a novel hazard is involved or the causal linkage between the putative hazard and the observed damage is unclear, screening assessments may be needed to establish susceptibility before proceeding to assess levels and probabilities of effects. This criterion is obviously situation-specific and will not be discussed further.

The seriousness of effects has been suggested as a criterion in other discussions of endpoints (e.g., AMS 1987) but is excluded here as inappropriate. This criterion includes severity, reversibility, and extent. If an endpoint has societal and biological significance then it should not be excluded simply because more serious effects are possible. Rather, both serious but low probability endpoints and less serious but potentially high probability endpoints should be assessed so that they can be considered and balanced in the risk management process.

2.3.2 Measurement Endpoints

Criteria for a good measurement endpoint are listed in Table 2-2. First, a measurement endpoint must correspond to or be predictive of an assessment endpoint. The environmental sciences literature is replete with examples of traits that were measured in the laboratory or field but which could not be explicitly translated into a societally or biologically important environmental value. If a measurement endpoint does not correspond to an assessment endpoint, it should be correlated with an assessment endpoint, or should be one of a set of measurement endpoints that predict an assessment endpoint through a statistical or mathematical model. For example, the assessment endpoint, landscape aesthetics, might be a function of of two measurement endpoints, a landscape dominance index and the percent of the landscape that is visibly disturbed.

Measurement endpoints should be readily measured. That is, it should be possible to quickly and cheaply obtain accurate measurements using existing techniques.

Measurement endpoints must be appropriate for the scale of the pollution, physical disturbance, or other hazard. It would be inappropriate to measure the outmigration of salmon smolts to determine the effects of an individual waste outfall but outmigration might be appropriate as a measure of the quality in an entire riverine watershed as fish habitat.

Measurement endpoints must be appropriate to the route of exposure. The organisms or communities that are measured should be exposed to the

Table 2-2. Characteristics of good measurement endpoints

1.	Corresponds to or is predictive of an assessment endpoint
2.	Readily measured
3.	Appropriate to the scale of the disturbance/pollution
4.	Appropriate to the route of the exposure
5.	Appropriate temporal dynamics
6.	Low natural variability
7.	Diagnostic
8.	Broadly applicable
9.	Standard
10.	Existing data series

polluted media and should have the same routes of exposure in approximately the same proportions as assessment endpoint organisms or communities. When such matching is not possible, then organisms that have the highest exposure should be used. For example, at sites where soil is contaminated, burrowing rodents have higher exposures than rodents that use surface runs and nests (McBee 1985). As another example, canopy trees have greater exposure to air pollutants than understory trees, and trees on ridge tops have high exposures to regional pollution while trees on the sides of ridges at the average inversion height have the greatest exposure to local pollutants.

Measurement endpoints should have appropriate temporal dynamics. If the hazard is episodic, then the measured response should be persistent so that evidence of effects will still be apparent after the event. For example, visible injury of leaves is apparent after air pollution episodes but photosynthetic rates recover rapidly.

Measurement endpoints should have low natural variability. Response that are highly variable among individuals or across space and time have low signal to noise ratios when used to measure pollution effects. As a result, either the effects are masked or large numbers of replicates must be used. For example, fecundity is more sensitive to most pollutants than mortality in fish, but fecundity is highly variable among individual females so fecundity effects are hard to distinguish in toxicity tests (Suter et al. 1987). The importance of variability depends on the relative scales of the variance and the measurements. For example, most environmental assessments address effects on the scale of years, so diurnal variance is irrelevant and variance due to climatic trends on the scale of hundreds to thousands of years is not detected.

It is desirable for measurement endpoints to be diagnostic of the pollutants of interest, to the extent that the pollutants have been identified. For example, concentrations of adrenal corticoids are indicators of stress in general, DNA single strandedness is indicative of genotoxins, and DNA adducts of benzo-a-pyrene (BAP) are indicative to DNA damage by BAP (McCarthy et al.)

It is desirable for measurement endpoints to be broadly applicable to allow comparison among sites and regions. For example, armadillos are probably good monitors of soil pollutants because they burrow and feed on soil and litter invertebrates. However, armadillos occur in a small portion of the United States, while mice of the genus Peromyscus are ubiquitous.

It is desirable for measurement endpoints to be standardized to allow precise comparisons among sites or tests. Standard methods and endpoints for toxicity testing are readily available for a variety of aquatic organisms, for some terrestrial animals, for a few plant responses, and for a few microcosms and mesocosms. Sources include the American Society for Testing and Materials (ASTM), American Public Health Association (APHA), Organization for Economic Cooperation and Development (OECD), and Environmental Protection Agency (EPA). Standard methods for measuring pollutant concentrations in the environment are available from the same organizations. Methods for biological monitoring are much less standardized and what standards exist (e.g., ASTM 1987) are not as widely used.

Finally, it would be desirable to use a measurement endpoint for which there is an existing time series of data so that background levels, variability, and trends can be estimated. There is the additional advantage that data from an ongoing monitoring or testing program is free. Potential examples are climatic data, air and water quality data, and harvest data for resource species.

2.4 POTENTIAL ASSESSMENT ENDPOINTS

Potential assessment endpoints for ecological risk assessments are listed in Table 2-3. They are divided into two categories, (1) traditional endpoints that have been used for local environmental risk assessments and may be useful in regional assessments and (2) endpoints that are characteristic of regions. The listed assessment endpoints are actually classes of endpoints; an endpoint for a real assessment would specify the entity and characteristic (e.g., frequency of kills of more than 100 fish of any species). Even at this level of generality, any list of endpoints will be incomplete. Anyone can imagine other assessment endpoints that may be useful in specific cases. The endpoints listed in Table 2-3 were chosen to have generic utility.

Table 2-3. Potential assessment endpoints for regional ecological risk assessments

<u>Traditional</u>

Population
- Extinction
- Abundance
- Yield/production
- Frequent gross morbidity
- Contamination (FDA Action Levels)
- Massive mortality

Community/ecosystem
- Market/sport value
- Recreational quality (e.g., eutrophication)
- Change to less useful/desired type

Abiotic
- Air and water quality standards

<u>Characteristic of Regions</u>

Species (population)
- Range

Productive capability
- Soil loss
- Nutrient loss
- Regional production

Pollution of other regions
- Pollution of outgoing water
- Pollution of outgoing air

Susceptibility
- Pest outbreaks
- Fire
- Flood
- Low flows

Landscape aesthetics

Climatic
- Continental glaciation
- Sea level rise
- Drought
- Increased UV radiation

2.4.1 Populations

Population-level assessment endpoints have generally been the most useful in local risk assessments because (1) responses at lower levels (i.e., organismal and suborganismal) have no social or biological significance; (2) populations of many organisms have economic, recreational, aesthetic, and biological significance that is easily appreciated by the public; and (3) population responses are well defined and easier to predict with available data and methods than are community and ecosystem responses. Clearly, the societal or biological significance of population-level responses depends on the societal or biological importance of the species. Changes of productivity of a soil nematode or a rotifer population would be unnoticed and unmourned by the public and would not have significant biological repercussions in most ecosystems. In the remainder of this discussion we will be referring to populations of socially or biologically important species.

The most drastic population level effect is extinction; it is well defined and potentially has great social and biological significance, particularly at regional scales. It should be predicted with good success if the hazard is habitat loss and with moderate success for toxic effects. Extinction can be monitored with relative ease for conspicuous species. If we declare a species functionally extinct when it is not sufficiently abundant to fulfill its societal or biological role (e.g., a fish that is too rare to support a fishery or a predator that is too rare to affect prey population size), all extinctions of macroorganisms are easily monitored. Extinction is a more useful endpoint at local scales than regional scales.

Although anthropogenic local extinctions are relatively common, regional scale extinctions are uncommon and represent a major failure of environmental management when they occur.

Abundance, production, and yield (harvestable production) are expressions of the ability of a population to fulfill a biological or resource role. If the yield of a resource population such as a timber tree or sport fishery declines, the societal significance is obvious. Abundance of nonresource species has societal importance if the species is missed. The biological significance of both abundance and production may be large or small depending on the natural variability of the species and its role in the biotic community. Abundance and production are well defined attributes. Although techniques exist to predict these quantitative population responses, the reliability of the techniques is not well established. Effects of habitat modification on wildlife can be predicted using the U.S. Fish and Wildlife Service's habitat evaluation procedure (Division of Ecological Services 1980) and effects of pollutants can be predicted by applying the effects observed in toxicity tests to population models for animals (Barnthouse et al. 1987, and in press) or plants (Larson and Heck 1984). Abundance is easily measured locally for many species but are difficult to measure over an entire region. Techniques exist for measuring production of most species in the field but they are more difficult and less accurate than abundance measures. For resource species, regional abundance or yield data are often available from resources agencies.

Frequent gross morbidity (tumors, lesions, and deformities) or mass mortality (fish kills and tree die-offs) are societally significant because they are aesthetically unappealing and because mortality diminishes the availability of resources. Morbidity and mortality are also significant because the public has come to interpret them as signs of pollution that may constitute a human health threat. Gross morbidities have little biological significance _per se_ but mass mortalities can be highly significant and have the advantage of being easily translated into monetary values (Economic Analysis, Inc. 1987). Mass mortality is relatively easily predicted if good exposure predictions are available because the most common toxicological endpoints represent laboratory mass mortalities (i.e., LC_{50}s and LD_{50}s). Gross morbidity is not presently predictable although deformities are observed in reproductive toxicity tests. Mass mortality of fish is readily apparent in inland waters and state agencies often keep a record of fish kills. Mass mortality of trees and coastal marine mammals are also apparent. Mass mortality of most other organisms is likely to go undetected. Even mass mortalities of birds in pesticide-sprayed fields and forests are likely to go undetected because of scavenging and the obscuring effect of vegetation (Balcomb 1986). Gross morbidity is more readily measured because the conditions persist and can be evaluated by inspection of a sample of organisms but has seldom been included in monitoring programs.

Contamination of populations by pollutants has societal significance if the organisms provide human food. This endpoint is well defined for many chemicals by the FDA action levels. It is readily predicted for aquatic organisms from concentrations in water and is relatively straight-forward for terrestrial plants, but the complexity of exposure in terrestrial wildlife (food, water, air, and soil can all be important) makes prediction of wildlife body burdens very difficult. Contamination is easily measured and is already monitored in commercial foods.

Population-level endpoints are appropriate to regional assessments under three circumstances. (1) If the subject of the assessment is a jeopardized species such as an endangered species or a declining species such as the black duck, then population endpoints must be evaluated at a regional scale where the region corresponds to the range of the species or the portion of the range where the decline is occurring. (2) Population-level endpoints are appropriate when the abundance or other characteristics of a species characterize the perceived value of a region. For example, preservation of old-growth forest in Northern California, Oregon and Washington has been an issue for decades but the issue has been largely expressed in terms of preservation of populations. First there was "save the redwoods" which meant save the oldest age classes of redwoods. More recently, saving the spotted owl has been an assessment endpoint that also expresses a desire to save the old-growth coniferous forest community type (Simberloff 1987). (3) Population-level endpoints could be used to characterize the state of a region by selecting a suite of species whose status would serve to integrate the physical and chemical disturbance of a region. These might include classic indicator species (e.g., sludge worms and may flies for polluted and clean aquatic environments, respectively), endangered or declining species, and commercial or recreational species.

2.4.2 Community/Ecosystem

Changes in the character of a biotic community can have major societal implications. If the market or sport value of a community changes, as from a fish community dominated by pelagic species such as lake trout or striped bass to one dominated by benthic species such as carp and suckers, the societal implications are obvious. Similarly, community changes such as severe eutrophication can diminish the recreational value of the community. Although there is a large body of literature on the economic value of recreation, the translation of environmental qualities into recreational utility is usually limited to complete loss of the resource as occurs when a beach is coated with oil (Economic Analysis, Inc. 1987). Changes of community type that do not directly involve commercial, sport, or recreational values are also likely to be regarded as changing the utility or desirability of the community. However, the definition of what constitutes a significant negative change in a community type is often ambiguous. A moderate increase in the trophic status of a lake may increase production of desirable fish species but diminish the value for swimming, boating, and aesthetic enjoyment, particularly in an oligotrophic lake like Lake Tahoe, California.

A change in community type is likely to have biological significance because large numbers of species and large areas are potentially involved. However, whether a particular change is biologically significant depends on the particular change and the community function evaluated. For example, conversion of a mixed forest to a pine plantation would decrease the number of animal species supported but could increase habitat for the endangered

red-cockaded woodpecker. The change in forest type would affect local hydrology by increasing transpiration and would have relatively small effects on retention of soil and nutrients.

Endpoints for most significant community transformations can be given good operational definitions. Examples include the conventional classification of lake trophic states and classifications of vegetation types.

Prediction of community changes due to physical disturbances (e.g., conversion of forest to pasture or filling of wetlands) is a trivial assessment problem if we know what types of communities will inhabit the sites. Effects on communities of additions of nontoxic pollutants (e.g., organic matter and nutrients) are reasonably predictable in aquatic systems and there is a growing body of information on sludge and waste water disposal in terrestrial systems that can provide a basis for prediction. Effects on communities of toxic chemicals are not directly predictable. They can be inferred from information on toxicity to component taxa and knowledge of the relationship between taxa (O'Neill et al. 1982, 1988; West et al. 1980; Dale and Gardner 1987) but there is not sufficient experience with this approach to evaluate its predictive power for community transformations. Microcosms and mesocosms are an alternate means of assessing toxic effects in communities. Because these experimental systems do not allow for long-term recovery, recolonization, and succession, they are more useful for assessing individual and population level effects in a realistic context than for assessing community transformations.

Community transformations that take the form of changes in vegetation are easily measured from satelite and aerial images or ground surveys. Monitoring changes in terrestrial animal communities and in aquatic communities requires greater effort in sampling or observation but present no conceptual problems.

At a regional scale, the appropriate expressions of community-level endpoints are frequency of changes of community type or changes in the area of community types. Examples include changes in the frequency of unacidified lake communities characterized by the presence of trout, and reduced area of old growth forests due to logging and fire. These may be assessed directly by characterizing the communities or, as mentioned above, indicator populations may be assessed. In the Adirondack lake example, one can predict the presence of a salmonid-dominated community by assessing effects of pH and aluminum on trout (Christenson et al. 1988) or by assessing landscape characteristics that lead to high exposures to acidity and aluminum (Hunsaker et al. 1986).

2.4.3 Air and Water Quality Standards

Although the derivation of air and water quality criteria and standards is a difficult and complex process, use of standards as assessment endpoints is simple. It is assumed that exceedence of standards is both societally and biologically significant. Standards are completely and precisely defined and measurable, and can be predicted by standard models of pollutant transport and fate. Their chief limitations are that they have no meaning outside the legal regulatory context and they only protect those environmental values that were included in the standard setting process. Sensitive responses such as reduction in plant growth by ozone or material damage by sulphates may be neglected in favor of human health. Poorly understood mechanisms (e.g., behavioral effects) are left out of the estimation of criteria and standards, and poorly understood effects (e.g., effects of acid deposition on trees) are left out of the standard-setting process entirely.

2.4.4 Regional Populations

The range of a population or species is the lowest-level endpoint that is useful for regional ecological risk assessment. Range is socially significant to people at the edge of a species' range who may lose the benefits of the species. Range reductions are biologically significant in that the functional properties of the species are lost in the former range and in that the species becomes more susceptible to extinction. Range is conceptually well defined and can be readily measured for macroscopic species. Range reductions that are due to local hazards that cause local extinctions are generally predictable, but range reductions due to a regional hazard (e.g., shrinkage of the range of a tree because of the combined effects of regional air pollution and suboptimal habitat at the periphery of its range) are not readily predicted.

2.4.5 Regional Productive Capability

Productive capability has clear societal significance, but that significance is discounted relative to current production. The potential biological significance of productive capability is also clear and, unlike the societal significance, is not mitigated by accounting procedures or human shortsightedness. Productive capability is difficult to define, predict, or measure, and realized regional production is a crude estimate of productive capability. The processes of soil and nutrient loss imply loss of productive capability if they exceed soil formation and nutrient input. Soil and nutrient loss can be readily measured in effluent rivers; losses in air are more difficult to measure but are much smaller in most of the United States. Production of resource species can be estimated from agricultural, forestry, and wildlife statistics. Prediction of soil loss is routine on the scale of individual fields, using the universal soil loss equation (Wischmeier and Smith 1978), but no good methods are available for predicting regional soil or nutrient loss. Prediction of agricultural and forestry production relies primarily on economic models rather than environmental models because of the importance of management and land conversion.

None of these indicators of productive capability is easily or reliably interpretable. The problem is in large part a matter of spacial aggregation and of aggregation of distinct processes. Soil export from a region does not indicate how soil might be moved around within a region, such as from fields to riparian lowlands or to the bottoms of reservoirs. Similarly, soil loss from agricultural fields has different implications than loss from a construction site that will no longer produce crops or forests. Erosion control at a construction site will improve water quality but has no implications for future terrestrial production. Increased nutrient export may reflect a loss of productive capability or may reflect increased fertilizer use and increased sewage disposal. Increases in realized production may reflect a genuine improvement in productive capability or simply more intensive management such as irrigation or conversion of mature natural forests to tree plantations. Conversely, some management practices such as herbicide use, selection of slow-growing varieties of street trees, or treatment of sewage to reduce nutrient content are intended to reduce total production. If productive capability is to be assessed, it may be necessary to address soil and nutrient loss at a smaller scale than a region or to address realized production in terms of specific valued species such as crops and timber trees, preferably normalized to acerage and input of fertilizer, water, and energy. Brown (1987) suggested ecologicly deflated production as an indicator of productive capability. It is calculated as realized agricultural production minus production from unsustainable practices such as tillage of highly erodible land or mining of ground water for irrigation. Clearly, endpoints for regional productive capability are still a subject for research.

2.4.6 Pollution of Other Regions

The amount of pollution exported by a region is an indicator both of damage done to adjoining regions and of the amount of pollutant chemicals in the regional environment. Pollution export is easily measured in outflowing rivers but not in air. It is predictable for both air and water for point sources and for those pollutants specified in effluent permits.

Pollution export is only crudely predictable for nonpoint sources and for noncriterion pollutants.

This endpoint is most useful and reliable as an indicator of relationships between regions such as pollution of estuaries by upstream regions. As an indicator of pollution of the exporting regions, it suffers from aggregation error due to retention of pollutants where they were deposited or transfer between compartments within a region. Thus, pollution export is insensitive in that it may underestimate or miss entirely an increase in regional pollution load.

2.4.7 Susceptibility

Pest outbreaks, property-damaging fires and floods, and stream flows that are inadequate to provide for dilution of effluents, consumptive uses, or navigation have clear societal and biological significance. Characteristics of a region can make it more or less susceptible to these events and those susceptibilities are potentially important regional assessment endpoints. These susceptibilities can be defined and measured in terms of frequencies of occurrence of events greater than a certain magnitude (e.g., fires burning more than 100 ha). It is much more difficult to predict how changes in a region will affect susceptibility although development of such capabilities is an active area of research.

2.4.8 Landscape Aesthetics

Although the aesthetic implications of changes in regional landscapes have social significance, they have no biological significance and no operational definition. Landscape aesthetics have been a subject of study, but they are difficult to measure or predict because of the critical role of culture, personal values, prior experience, and training. The Englishman's pleasure in a "land parceled and pieced" contrasts with the westerner's love of "wide open spaces". Aesthetics may even be in conflict with biological values. The general aversion to swamps has contributed to their destruction and clean clear-cuts are generally preferred to those with the slash left in place to retain nutrients and retard erosion. In sum, landscape aesthetics is not a useful generic assessment endpoint, but it may be useful in specific instances where there is a consensus on the aesthetic implications of an action.

2.4.9 Climatic

In the last two decades, concerns have been raised about modification of the global climate or regional climates by fossil fuel combustion, release of chlorofluorocarbons, release of particulates, nuclear war, deforestation, and devegetation. These could cause glaciation, sea level rise, drought, or biological damage by ultraviolet radiation; endpoints that have greater social and biological significance than any of those previously discussed. These endpoints are obviously measurable but are sufficiently severe that waiting until effects can be determined by measurement is not a desirable assessment approach. Prediction of regional and global climatic effects is a major activity, but the validity of the models used is questionable. The implications of climatic change are grossly predictable by identifying the communities that occur now or occurred in the past in areas that have or had climates similar to the predicted climate. Although agriculture and commercial forestry are relatively adaptable, the modern circumstance of isolated fragments of natural communities precludes the assumption that communities or species will move to their appropriate habitats.

2.5 MEASUREMENT ENDPOINTS

Potential measurement endpoints for regional risk assessment are listed in Table 2-4. As with the assessment endpoints, they are divided into those that are traditional and those that are characteristic of regions. As with the assessment endpoints, these are classes of endpoints. For example, actual measurement endpoints for individual mortality include median lethal dose, the lowest dose at which no deaths occurred, and the number of dead individuals observed following a pollution episode. It is more difficult to generalize about the utility of measurement endpoints than about assessment endpoints because the ability to measure an environmental characteristic, and its relation to characteristics of the hazard are situation specific.

2.5.1 Individual

The endpoints of nearly all laboratory toxicity tests are summaries of responses of individual organisms. For example, the LC_{50} is a statistical estimate of the concentration at which the median individual dies. Death,

Table 2-4. Potential measurement endpoints for regional ecological risk assessments

Traditional	Characteristic of Regions
Individual	Landscape descriptors
Death (LC_{50})	Fractal dimension
Growth	Contagion
Fecundity	Dominance
Overt symptomology	Diversity
Biomarkers	Area of ecosystem/use
classes	
Population	Rate of movement of ecotones
Occurrence	Length of ecotone/edge
Numbers/density	Species (populations)
Age structure	Range
Reproductive performance	Material export
Yield/production	Soil export
Frequency of gross morbidity	Nutrient export
Community	Pollutant chemical export
Number of species	Susceptibility
Species evenness	Frequency of pest outbreaks
Species diversity	Frequency/area of fires
Market/sport value	Frequency/severity of floods
Saprobic index	Frequency/severity of low
flows	
Other indices	Hydrologic variables
Ecosystem	Regional production
Biomass	
Productivity	
Nutrient export	
Abiotic	
Pollutant concentrations	
Physical state variables	
(TSS, TDS, DO, Temperature)	

reproduction, and growth can be related to population and ecosystem-level assessment endpoints through the use of population and ecosystem models (Sects. 2.4.1 and 2.4.2). In addition, regulatory agencies have developed safety factors for interpretation of these standard test endpoints (e.g., Urban and Cook 1986). Overt symptomology (visible effects such as spinal deformities in fish and chlorosis of plant leaves) and biomarkers (biochemical, physiological, and histological indicators of exposure or effects) are potentially diagnostic. Handbooks are available for attributing visible plant injury to specific pollutants (Jacobson and Hill 1970, Malhotra and Blauel 1980) and many biomarkers are diagnostic of classes of chemical (e.g., metalothionines for metal exposure) or for specific chemicals (e.g., DNA adducts of specific mutagenic chemicals) (McCarthy et al. in press). Overt symptomology and biomarkers, as well as behavioral responses, currently can not be used to predict assessment endpoints even though they have clear implications for the health and survival of organisms. There are currently no models that relate symptoms, biomarkers, or behavior to higher level effects. In general, individual responses are difficult to measure in the field, but there are obvious exceptions such as responses of individual trees.

2.5.2 Population

The conventional population parameters (occurrence, abundance, age structure, birth and death rates, and yield) are poor subjects for laboratory tests but are popular components of ecological field studies. They are directly interpretable in terms of assessment endpoints for valued populations. Occurrence and abundance are easily measured, but age structure is difficult to establish for many species. Birth rates, death rates, and yield are difficult to establish for many species (but not annual plants). The scale of population responses of large vertebrates is appropriate for regional risk assessments. Population responses have good temporal dynamics in that they integrate chronic and acute exposures. Their variability depends on the species. They are not diagnostic. Methods for population surveys are not standardized but there are generally accepted methods applicable to most species.

The frequency of mass mortalities and the frequency and nature of overt morbidity correspond to assessment endpoints. Overt morbidity is readily measured in the field for most vertebrates and mass mortalities are noted by many local and state agencies. Frequencies of overt morbidity are quite variable and care must be taken in diagnosis of lesions and tumors to distinguish effects of parasites or mechanical injury. These endpoints are not standardized and, with the possible exception of fish kills, are unlikely to have existing data.

2.5.3 Community

The most commonly used community characteristics in environmental monitoring are the number of species, species evenness, and species diversity. They are popular because they conveniently summarize the data generated by biotic surveys. They are easily measured for macroorganisms and temporally integrate acute and chronic exposures. For most macroscopic flora and fauna they have reasonably low variance, but the evenness and diversity of invertebrates tend to be highly variable. Commuity endpoints are broadly applicable but not diagnostic or well standardized although some standards for community sampling exist (APHA 1985, ASTM 1987). The problem comes in relating these numbers to assessment endpoints. If the nature and aspect of the community has not been affected, then changes in number, evenness, and diversity must be interpreted in terms of the species that have appeared, disappeared, or changed in relative abundance as a result of the presence of the waste. In other words, the effects must be assessed at the population level because the number and diversity of species is no longer believed to confer stability or any other value on the community. Certainly the increase in species number and diversity that results from colonization of disturbed areas by weedy species is not valued or of great consequence. If the nature and aspect of the community has been changed, then number, evenness, and diversity numbers are simply adjuncts to the description of the changed community type.

Another type of community-level endpoint is indices of community quality, which may be indicative of pollution effects or of habitat quality in general. The best example of a community pollution index is the saprobic index (Hynes 1960). This index arrays aquatic communities with respect to conventional organic pollution (i.e., sewage and similar effluents) which predictably replace one set of species with another. They are unlikely to be useful at waste sites and it is unlikely that useful new pollution indices can be devised for waste sites because wastes are unlikely to have a suitably stereotypical effect. Indices of generic community quality, such as the index of biological integrity (IBI)

(Karr et al. 1986), show promise as indicators of the state of communities. The IBI provides an indication of the physical and chemical quality of streams based on the species composition, trophic composition, abundance, and condition of fish. Community quality indices, like diversity indices, are statistically intractable and greatly reduce the information obtained from a biotic survey by reducing it to one number. However, if an index is well characterized for a region as the IBI is for the north central states, it can be used to indicate how far communities have diverged from an undisturbed state. For most regions and community types, appropriate indices and baseline data are not currently available.

The indicator species concept is a reduced form of the community index. The presence or abundance of a species that is thought to be either pollution sensitive or tolerant is used to indicate the status of a community. Like the saprobic index, indicator species have been effective in assessing oxygen-demanding pollution but not for other types. Therefore, an indicator species can not reliably define effects of waste sites.

To be relevant to regional assessments, community responses need to be scaled-up to the regional level. The community properties and indices discussed above are intended to characterize a particular site. Regional assessments need a measure of the state of the individual community types in the region or a means of integrating measurements from individual sites. These measures could be as simple as percentages of sites below some threshold value (e.g., streams with fewer than 3 species of fish or forests less than 100 years old), but more sophisticated measures can be easily imagined. The chief limitation is the lack of consistent measurements of community properties from sites distributed across a region.

2.5.4 Ecosystems

Ecosystem properties relate to the exchange of energy and nutrients among functionally defined groups of organisms and between organisms and the environment. The most commonly measured ecosystem properties are biomass of the system or its components (e.g., trophic levels), productivity of the system or its components (e.g., primary and secondary production), and nutrient dynamics (i.e., rates of elemental cycling and loss). These do not correspond to any assessment endpoint but all relate to productive capability. In particular, the realized productivity of an ecosystem is an estimator of its productive capability. Ecosystem properties tend to vary with climatic conditions, and are not diagnostic, but they are broadly applicable. There are no standard methods for measuring toxic effects on ecosystem processes in the field, but the EPA has recently adopted laboratory microcosm protocols that include some measurements of ecosystem processes (Office of Pesticides and Toxic Substances 1987).

Properties of individual local ecosystems like those of communities must somehow be related to a regional scale. The potential approaches would be the same, and consistent data from the ecosystems in a region is equally lacking. In addition, the individual ecosystem properties have no inherent social value and must be interpreted in terms of the ability to produce resources, sustain desired community types, or other assessment endpoints.

2.5.5 Abiotic

Measurements of pollutant concentrations, pH, dissolved oxygen, suspended solids, and other abiotic properties of environmental media are readily performed and there are standard procedures for many analyses. They serve as endpoints for those chemicals for which there are air or water quality criteria (i.e., if a criterion or standard is the assessment endpoint then ambient concentrations are the measurement endpoints). For non-criterion chemicals, the endpoints must be some effect which is then associated with predicted or measured concentrations.

2.5.6 Landscape Descriptors

The landscape descriptors produced by the new and growing field of landscape ecology (O'Neill et al. 1988, Forman 1986) are appealing as potential measurement endpoints because they describe characteristics of a region as a whole. They are relatively readily measured thanks to the abundance of high quality satellite and aerial imagery, and to recent advances in image analysis and geographic data analysis. They also have low natural variability, are broadly applicable, and historic aerial photos may allow extension of a landscape data series back for 40 years. However, efforts are only beginning to relate them to assessment endpoints. For example, Franklin and Forman (1987) modeled the effects of clearcutting pattern on landscape descriptors (length of edge, patch size, and proportions of uncut, cut, and interior uncut) and on the amount of nonhuman forest harvesting (fire, blowdown, insect and fungal outbreaks, and landslides). Another example is the attempt to relate abundance of wildlife, particularly birds to patch size (Freemark and Merriam 1986, Orians 1986), to relative amounts of edge and interior (Kroodsma 1984a, 1984b), and to the availability of corridors between patches (Henderson et al. 1985). The proportion of a landscape disturbed by human development is more comprehensible to the public than other landscape descriptors and has been used as an assessment endpoint (e.g., Walker et al. 1987), but like the other measurement endpoints in this class, it should be related to some regional value or utility. All of these landscape descriptors have been designed to quantify physical disturbance in the terrestrial environment; their applicability to toxic effects and aquatic ecosystems is problematical.

2.5.7 Species/Populations

The range of a species or population is an intrinsically regional measure and corresponds to the assessment endpoint discussed previously (Sect. 2.4.4). Ranges are known for game species, birds, fish, trees, and most other species that would be useful for assessment endpoints. The range of a species usually has low variability and determinations of changes in range can often draw on existing data series. It is applicable to hazards that encompass all or most of the range of a species or of a spatially distinct population.

2.5.8 Material Export

Export of materials relates to the productive potential of a region (Sect. 2.4.5) and pollution of other regions (Sect. 2.4.6). It is readily measurable in water and the natural variablility is primarily due to climatic and hydrologic factors which can be corrected for. It is diagnostic for anthropogenic pollutants, is broadly applicable, measurement methods are standardized, and existing data sets can be used.

2.5.9 Susceptibility

Use of frequencies and intensities of pest outbreaks, fires, floods, and low flows to estimate susceptibility of a region to these events amounts to regional scale epidemiology. The problems are the same as in prospective epidemiology, using small samples of past events to estimate the probability of future events. The samples are small because the frequencies of severe events are low, making it difficult to reliably detect changes in frequencies resulting from regional change. The solution is to develop regional indicators of susceptibility to severe events. For flood and low flows this is a matter of extrapolating to extreme events the hydrologic parameters that describe the retention of water by a watershed. For example, Gosselink and Lee (1987) suggested assessing the effects of lost riparian wetlands on flooding by using the heights of discharge curves and the water residence times (stored volume at flood stage/discharge at flood stage). Additional parameters are needed to describe the role of uplands in water control (USFS 1980). Fire susceptibility is predictable from species composition, fuel loads, and dryness. It is not clear what would be measured to describe the susceptibility of a region to pest outbreaks.

2.5.10 Regional Production

As discussed previously (Sect. 2.4.5), realized regional production can be used as a crude indicator of productive capability. Crop and forest production statistics can be obtained from the USDA Crop Reporting Service and Forest Service. These are accurate, free, provide long data series, and have general applicability. Primary production of other community types must be estimated from assumptions and literature values (Turner 1987c) but these estimates are not useful for assessment since assumptions and literature values do not respond to hazards. Monitoring programs to determine the production of communities other than crops and forests would be very expensive relative to the utility of the data in regional risk assessments. Use of only the crop and forest data to estimate regional production would be obviously incomplete and aggregating crop and forest yield as an estimate of regional production would simply obscure the responses of the individual crops, forest tree species, and forest community types.

2.6 ASSESSMENT GOALS AND ASSESSMENT ENDPOINTS

It is not always possible for an assessment endpoint to satisfy all of the criteria in Table 2-1 and it is nearly impossible for a measurement endpoint to satisfy all of the criteria in Table 2-2. The relative importance of the criteria depends in part on the type of assessment. Three general goals of regional assessments are discussed below: explanation of observed effects, evaluation of actions with regional implications, and evaluation of the state of a region.

2.6.1 Explanation of Observed Regional Effects

Certain regional scale environmental effects are observed before their causation is understood. Examples include the decline of the peregrine falcon, the acidification of lakes in the northeastern United States, and the decline of high elevation forests in the Appalachians. In these cases, the purpose of assessment is to establish causation and the assessment endpoint is provided by the assessment topic. The measurement endpoints must have close causal links to the assessment endpoint and must be diagnostic of the mechanism involved in at least one causal link. They must also have appropriate spatial scale and temporal dynamics. Examples include body burdens of xenobiotic chemicals in falcons that fail to reproduce and body burdens corresponding to no observed effect levels in reproductive toxicity tests. On the other hand, it is not particularly important that measurement endpoints be easily and cheaply measured. When a serious problem is known to exist there is public support for spending money on measurement and a program focused on a single problem can expend more effort and money on each of a few pertinent measurements. Similarly, broad applicability, use of existing standard methods, and use of methods that have generated existing data are less important than standardizing the measurements that are most applicable to assessing the identified problem.

2.6.2 Evaluation of an Action with Regional Implications

Another goal of regional assessment is predicting the regional implications of environmental decisions. Examples include (1) licensing a pesticide for use on corn that can be expected to be used at approximately the same time on thousands of fields all across the corn belt and (2) permitting a new sewage outfall in a river that is already subject to anoxic conditions during low flows. The assessment endpoints in these cases are likely to be scaled-up versions of the endpoints used in local-scale assessments. For example, in a local assessment of a new pesticide an assessment endpoint might be the expected number of birds killed or probability that birds will be killed, whereas a regional assessment would use effects on the abundance of a regional avian population. In most cases no new measurements would be available for regional assessments so the same measurement endpoints would be used in an assessment model with a regional scope. In the pesticide example, the same avian LD_{50} or field test results as are used in local-scale assessments would be used in a model of avian population dynamics in a regional-scale mosaic of habitats some of which are being sprayed.

In some cases, regional effects of decisions are not simply scaled-up local effects. A conspicuous example is the transformation of pollutants from numerous individual sources into new regional pollutants, including generation of ozone and PAN from hydrocarbons and oxides of nitrogen and generation of sulfate aerosols from local SO_2 emissions. Such emergent properties of regional-scale disturbances often are not predicted but they may be detected by assessments of changes in the state of regional environments.

2.6.3 Evaluate the State of a Region

A third purpose of regional assessment is to evaluate the state of regions so as to (1) determine whether regulatory actions are improving environmental quality or (2) determine whether some hazard that is not being addressed is having environmental effects. In the first case, the assessment program is a validation of the regulatory assessments so the assessment endpoints that were used in the regulatory actions should be used in the validation. Measurement endpoints should be clearly representative of those assessment endpoints, should be sensitive to the hazard being regulated and should be readily measured, broadly applicable, and standard so that the effectiveness of actions can be evaluated in a comparable manner at sites within and among regions. In the second case, it is desirable to consider all endpoints so that nothing will be missed. It is obviously impossible to monitor everything adequately, but the number and severity of surprises can be minimized.

The development of endpoints for assessing the state of regions constitutes a difficult research problem. As Dayton (1986) points out, most purely observational studies have had little utility because the complexity of causal factors creates variance that is perceived as noise. This noise results in a high probability of type II error (i.e., missing real effects) which is particularly difficult to overcome if the goal is to detect sensitive early indicators of effects. As a result, "credible early warning signals have been the succubus of most pollution workshops..." (Dayton 1986). One possible solution is to use multiple types of endpoints with complementary qualities. One type of measurement endpoint would be summarizations of data from existing environmental, resource, and economic monitoring programs. Because the data are essentially free and are likely to cover a variety of species and other relevant regional characteristics, they need not be perfectly appropriate but they must be reasonably well standardized and should not have extreme natural variability at the time scales of interest. A second type of measurement endpoint addresses specific areas or entities within a region that have regional importance and that are thought to be particularly vulnerable to a broad class of pollutants or other hazards. Sensitive, low variance measurement endpoints with appropriate spatial and temporal scales might be used in those locations. For example, a variety of persistent hydrophobic pollutants accumulate in the sediment of estuaries so their effects might best be monitored in benthic organisms with a suite of biomarkers such as the alkaline unwinding assay that are not chemical specific but indicate a mode of toxic action. Another example might be movement of ecotones (boundaries between types of communities) in response to climatic change or similar stress. Finally, if endpoints can be developed that integrate stress on a region, even if they are not terribly sensitive or diagnostic, they could be used as a general warning that something is changing. For example, frequency of observed fish kills is a rather crude example of an indicator of the general water quality in a river basin.

The difficulty of regional monitoring of environmental quality is reflected in the fact that tens of millions of dollars spent on monitoring the chemical quality of surface water in the U.S. has not satisfactorily answered questions about trends or the efficacy of current regulatory strategies (GAO 1986, NRC 1987). Biological monitoring and terrestrial monitoring present additional serious challenges.

2.7 CONCLUSIONS

Because the term "endpoints" has been used to describe the numeric results of toxicity tests and field monitoring programs as well as to describe the object of an environmental assessment, measurement endpoints have been used as de facto assessment endpoints. Once the distinction between these endpoints is made, it becomes clear that the object of environmental risk assessments is not to predict the probability of occurrence of fathead minnow LC_{50}s in rivers or of changes in fractal dimensions of landscapes. A major task of risk assesssors is extrapolating from these measurement endpoints to the assessment endpoints (e.g., to fish abundance in rivers or the productive capability of a region). In many cases the necessary extrapolation models do not exist, and in some cases the conceptual bases for such models do not exist. Regional-scale measurements and indices need to be related to regional values. Methods need to be developed to estimate effects on regional populations and communities from toxicity test endpoints and measured local effects.

Relationships need to be developed between body burdens, biomarkers and other symptomology used in biological monitoring programs and effects on populations. These needs must be met by new research.

Regional risk assessments have a particular need for data bases of spatially and temporally extensive and consistent measurement endpoints. Long time series of data are particularly valuable and difficult to come by. All of the assessment goals described previously could be enhanced if regional risk assessors could consider trends rather than regional snap shots. In particular, if trends in regional state variables were assessed then deterioration in environmental values could be identified earlier than if the deterioration must be apparent in temporal isolation. Unfortunately, few monitoring programs are sustained beyond a few years and environmental data bases often are not sustained and updated after they are created. Regional risk assessment is particularly dependent on sustained commitment from the responsible agency.

Finally, regional-scale assessment endpoints are much less readily identified than are local-scale endpoints. At regional scales it is particularly apparent that although we care about all components of the environment we simply can not keep track of them all. In addition, the relative utility to regulators of the various whole region descriptors that are being developed is not apparent without guidance concerning the regulators' values. Therefore, it will be important for risk assessors and risk managers to identify the regional values that have greatest importance so that efforts can be directed to developing the data and assessment tools needed to assess risks to those endpoints.

3. DEMONSTRATION OF A REGIONAL RISK ASSESSMENT

3.1 INTRODUCTION

The demonstration addresses the impact of ozone and resultant insect outbreaks on land cover, wildlife habitat, and water quality in the Adirondack region of New York. The objectives are

1. to explore probabilistic methods for spatial, regional models;
2. to test an approach for assessing regional ecological risk;
3. to evaluate the sensitivity of disturbance effects to initial landscape pattern; and
4. to demonstrate the linkage between terrestial and aquatic systems in regional risk analysis by linking a model of land cover change to a water quality model.

Figure 3-1 outlines this approach. Because our purpose is to demonstrate a regional risk assessment and no resources were available for data collection, we chose a region (the Adirondacks) with available data and models. The simulated disturbance is biologically plausible but unlikely to occur in the Adirondack region.

3.2 CONSTRUCTION OF DEMONSTRATION AND DESCRIPTION OF DISTURBANCE PHENOMENON

We selected ozone, a regional air pollutant, as the hazard. Data exist on ozone concentrations and distributions, and there is considerable information on its effects on terrestial ecosystems. In the heavily forested Adirondacks, the immediate effect of elevated ozone concentrations would be physiological stress in conifers (Fig. 3-2). Conifers are much

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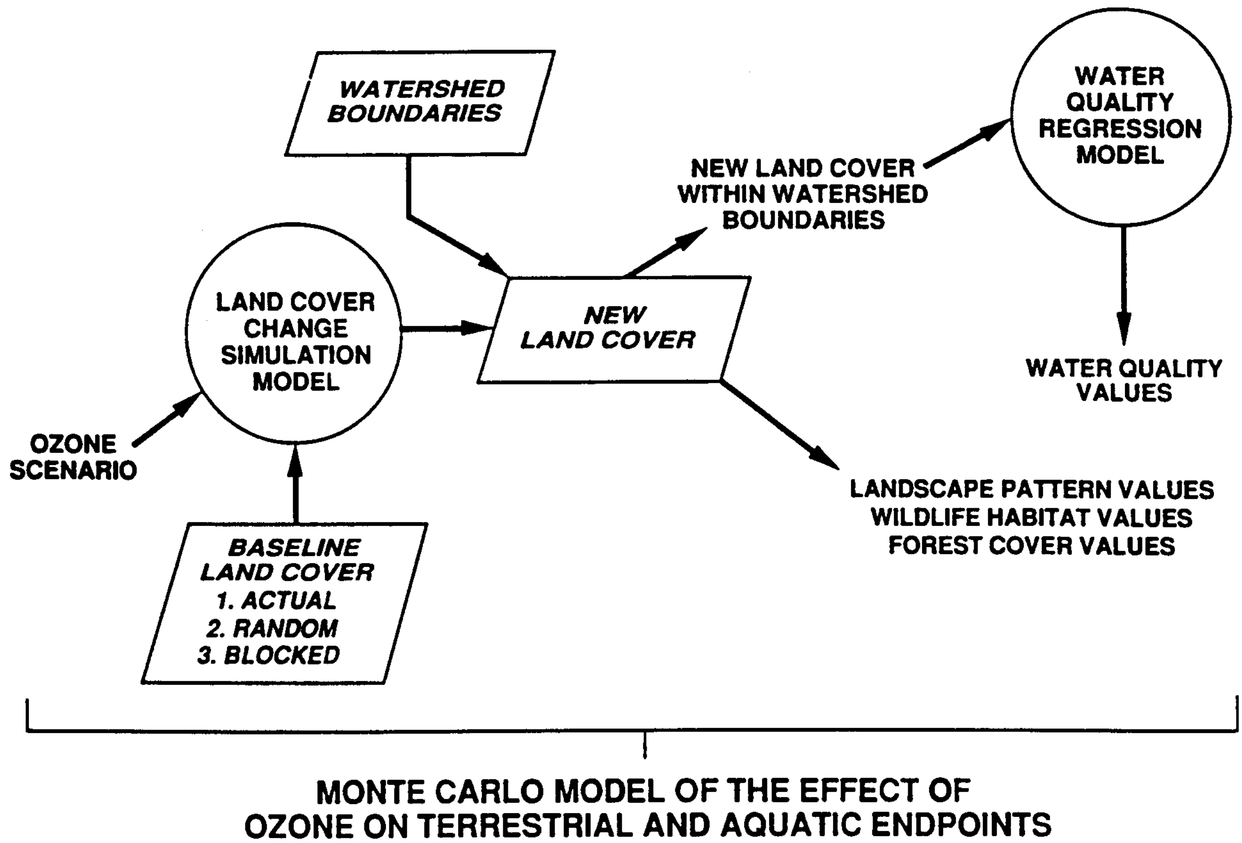

Fig. 3-1. Structure of Monte Carlo model of the effect of ozone on terrestrial and aquatic endpoints.

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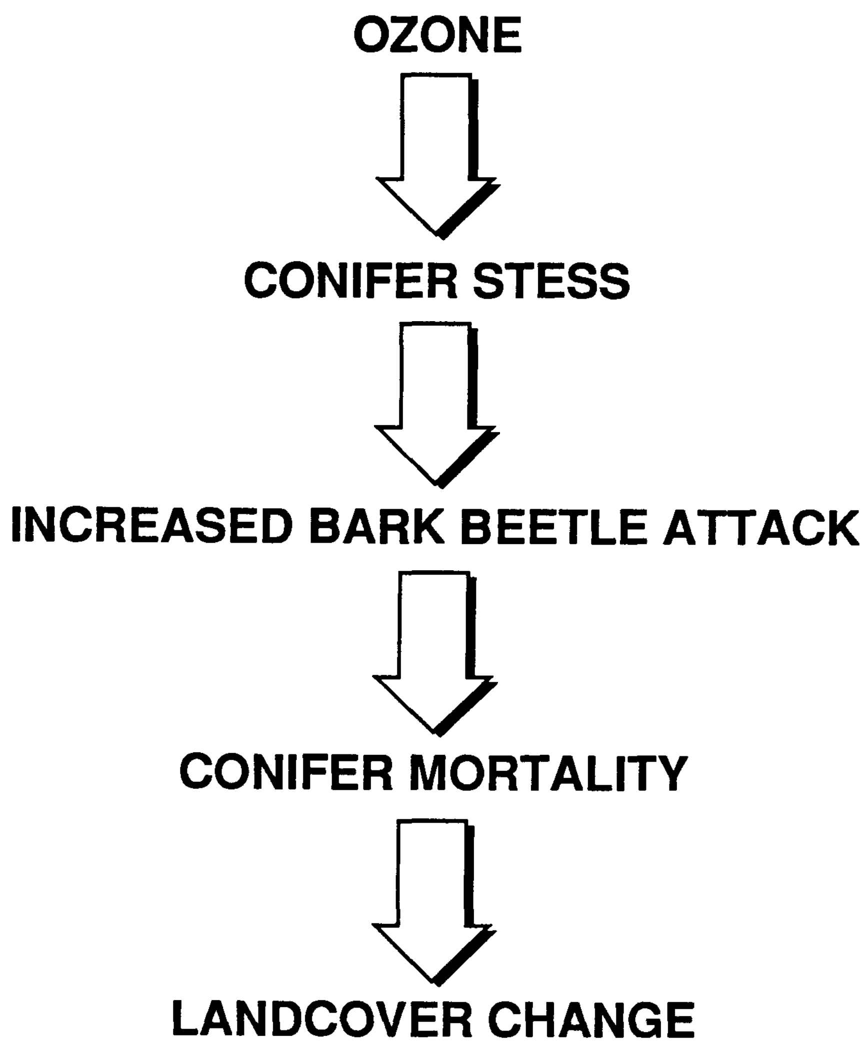

Fig. 3-2. Mechanism of the hypothesized environmental disturbance.

more sensitive to ozone than are hardwoods and exhibit stress symptoms at ozone concentrations considerably below those required to produce direct tree mortality (Heck et al. 1986, USEPA 1986b, Smith 1981).

Ozone stress can lead to an increase in the frequency and intensity of bark beetle attacks in conifer stands (USEPA 1987, 1986; Smith 1981; Stark et al. 1968; USDA 1980). Bark beetles are an indigenous feature of forested landscapes in the United States. Trees that are physiologically stressed are more susceptible to attacks. Once a tree has been successfully attacked, some species of bark beetles will then attack neighboring trees. In this manner, trees can be killed in patches that range in size from 1 to >50 ha (Thatcher et al. 1980, Stark et al. 1968). Although the bark beetles indigenous to the Adirondack region have not displayed this type of patch kill, we use this patch scenario in our demonstration to illustrate a spatially heterogeneous impact induced by a regional hazard.

Because of its spatial patterning, beetle-induced conifer mortality can cause impacts beyond simply altering the amount and type of forest cover. The loss of conifer cover can affect lake water quality because coniferous vegetation tends to acidify soils and, subsequently, lakes (Brady 1974). It is reasonable to expect that destruction of coniferous forest within a watershed would result in a decrease in lake water acidity. The patchy quality of bark beetle-induced conifer mortality could also change the amount of forest edge and interior forest habitat by dissecting the forest cover of the region. This in turn could affect wildlife in the region.

For the demonstration, we selected a wide array of ecological endpoints that might be affected indirectly by ambient ozone concentrations (Table 3-1). The endpoints were selected to be of ecological or economic concern and to vary in their innate sensitivity to land cover pattern.

Although risk assessment traditionally has evaluated negative environmental impacts, in regional assessment some measurement endpoints may relate to positive assessment endpoints (see Chapter 2). The pH shift endpoint could be negative or positive, and the acid improvement endpoint represents a positive effect. Also, while a decrease in one edge habitat would have a negative impact on a wildlife species dependent on that edge habitat, the associated increase in another edge habitat could be positive for a different species. We evaluated both pattern-sensitive and pattern-insensitive endpoints to ascertain whether spatial modeling was necessary to quantify regional environmental risks.

Because bark beetles are distributed randomly within a susceptible forested landscape, one can only state with a given probability where an attack will occur. Therefore, our approach was to impose a uniform level of ozone stress across the region and then randomly impose the bark beetle attacks. Monte Carlo simulation was used to examine 100 different arrays of beetle attacks distributed randomly over the landscape. Different arrays of bark beetle attacks changed the risk of a significant change in lake water quality because water quality for a given lake only changed if a bark beetle attack happened to occur within its watershed and significantly changed the percentage of the watershed in conifers.

A unique aspect of the demonstration was the linkage of spatial modeling with Monte Carlo techniques. Although previous risk assessments

Table 3-1. Ecological endpoint measures used in Adirondack demonstration

Measure	Definition
LAND COVER ENDPOINT MEASURES	
Forest	Percent of region classified as forest
Deciduous	Percent of forest classified as deciduous
Conifer	Percent of forest classified as coniferous
Mixed	Percent of forest classified as a mixture of conifer-deciduous trees
EDGE HABITAT ENDPOINT MEASURES	
Deciduous-open	Kilometers of deciduous forest bordering open land (agricultural, urban, wetland,barren, or shrubland)
Coniferous-open	Kilometers of coniferous forest bordering open land
Mixed-open	Kilometers of mixed forest bordering open land
Deciduous-agriculture	Kilometers of deciduous forest bordering agriculture
Coniferous-agriculture	Kilometers of coniferous forest bordering agriculture
Mixed-agriculture	Kilometers of mixed forest bordering agriculture
Deciduous-wetland	Kilometers of deciduous forest bordering wetlands
Coniferous-wetland	Kilometers of coniferous forest bordering wetlands
Mixed-wetland	Kilometers of mixed forest bordering wetland
FOREST INTERIOR ENDPOINT MEASURE	
Forest interior	Total amount of forest land (ha) further than 200 m from any nonforest land
LANDSCAPE INDICES ENDPOINT MEASURES	
Dominance	Degree to which the region as a whole is dominated by one or two land cover types
Contagion	Degree to which land cover types are grouped within the region
LAKE WATER QUALITY ENDPOINT MEASURES	
Lake pH shift	Percent of headwater lakes which experience a pH shift greater than or equal to 0.2 pH units
Acid improvement	Percent of lakes with a pH greater than 5.5 which initially had a pH $\leq$ 5.5

have used Monte Carlo modeling techniques (Barnthouse and Suter 1986, Barnthouse et al. 1987, O'Neill et al. 1982, Suter et al. 1984), few if any have used spatial models. Likewise, few spatial models have used probabilistic functions (but see Browder et al. 1988, Turner 1987c). Spatial modeling allowed us to capture the effect of spatial pattern on the regional endpoints, while Monte Carlo techniques allowed us to quantify risk.

The linkage of terrestrial and aquatic effects was also essential to our objectives. Local assessments generally focus on one aspect of the environment. Such an approach may be appropriate for local risk assessments, but regional assessments must consider the effects of changes in one ecosystem on other aspects of the environment. Obvious examples of this linkage between terrestrial and aquatic ecosystems are nonpoint-source water pollution problems, acid rain, and loss of wetlands.

3.3 METHODS USED IN DEMONSTRATION

Regional risk assessment has two phases--the problem definition and the problem solution (Figure 1-1). In the problem definition phase, the reference environment is defined, the endpoints are selected, and the source terms for the hazard are developed. Exposure assessment and effects assessment occur in the solution phase.

3.3.1 Reference Environment

Actual landscape. Land use and land cover (LUDA) data from the U.S. Geological Survey (USGS) for a portion of the Adirondack State Park were used to define the actual baseline land cover (USGS 1983). The region encompassed

308,408 ha. The data were in raster format; each grid cell encompassed 4 ha and was assigned one of the eight land cover catagories shown in Table 3-2.

Sixty-six headwater watersheds within the region were used to examine the impact on lake pH. A digitized polygonal file of the watershed boundaries (Hunsaker et al. 1986) was rasterized into 4-ha grid cells and overlaid on the land cover data. Each grid cell was assigned a watershed identifier (i.e., the number of the watershed if the cell was located within one of the 66 watersheds or a 0 value if the cell was located outside). The watersheds ranged in size from 16 to 4164 ha and altogether encompassed 7% of the total study area.

Altered Landscape. We developed two altered baseline landscapes to evaluate the sensitivity of disturbance effects to initial landscape pattern. The altered baseline landscapes have essentially the same percentage of each land cover type as the actual baseline landscape (Table 3-3). The number and location of water cells were kept constant in all baseline landscapes. In the random landscape, land cover was randomly distributed across the region. Cells of the same land cover were grouped together for the blocked landscape (Figure 3-3). For most landscape values the random baseline landscape is similar to the actual LUDA baseline landscape, whereas the blocked baseline landscape differs dramatically from the actual for measures of edge and contagion. In the blocked landscape, contagion is close to a maximum value for the specified conditions (18.62). For 94 landscapes with six or seven land use types, the contagion index ranged from 9.5 for the Boston area to 22.8 for Lewiston, Maine (O'Neill et al. 1988).

Table 3-2. Land cover types in the Adirondacks

Type	Definition
Urban	Cities, towns, roads, built up areas
Agriculture	Areas of cropland and pasture, includes orchards
Water	Open bodies of water greater than 4 ha in size
Coniferous forest	Area occupied by coniferous tree species
Deciduous forest	Area occupied by deciduous (hardwood) tree species
Mixed forest	Area occupied by a mixture of coniferous and deciduous trees
Wetland	Swampy or boggy areas, excludes coniferous swamps and bogs
Barren	Quarries, rock outcrops, sand

Table 3-3. Landscape values for the three baseline landscapes

	Actual	Random	Blocked
COVER (%)			
Forest	93.6	92.2	93.6
Deciduous	39.8	36.3	38.0
Coniferous	21.6	24.1	21.9
Mixed	38.6	39.5	40.1
Urban	1.6	1.8	1.6
Agriculture	2.0	2.2	2.0
Water	2.2	2.2	2.2
Wetlands	0.5	1.4	0.5
Barren	0.1	0.2	0.1
EDGE HABITAT (km)			
Deciduous-open	67	163	23
Coniferous-open	143	216	4
Mixed-open	152	281	8
Deciduous-agriculture	15	53	16
Coniferous-agriculture	73	89	0
Mixed-agriculture	57	102	0
Deciduous-wetlands	15	45	0
Coniferous-wetlands	10	36	0
Mixed-wetlands	15	70	8
FOREST INTERIOR (ha)	260,056	241,816	273,300
LANDSCAPE INDICES			
Dominance	1.47	1.40	1.47
Contagion	14.00	13.62	18.62
LAKES			
Number of lakes	66	66	66
Number lakes with pH ≤ 5.5	8	5	10

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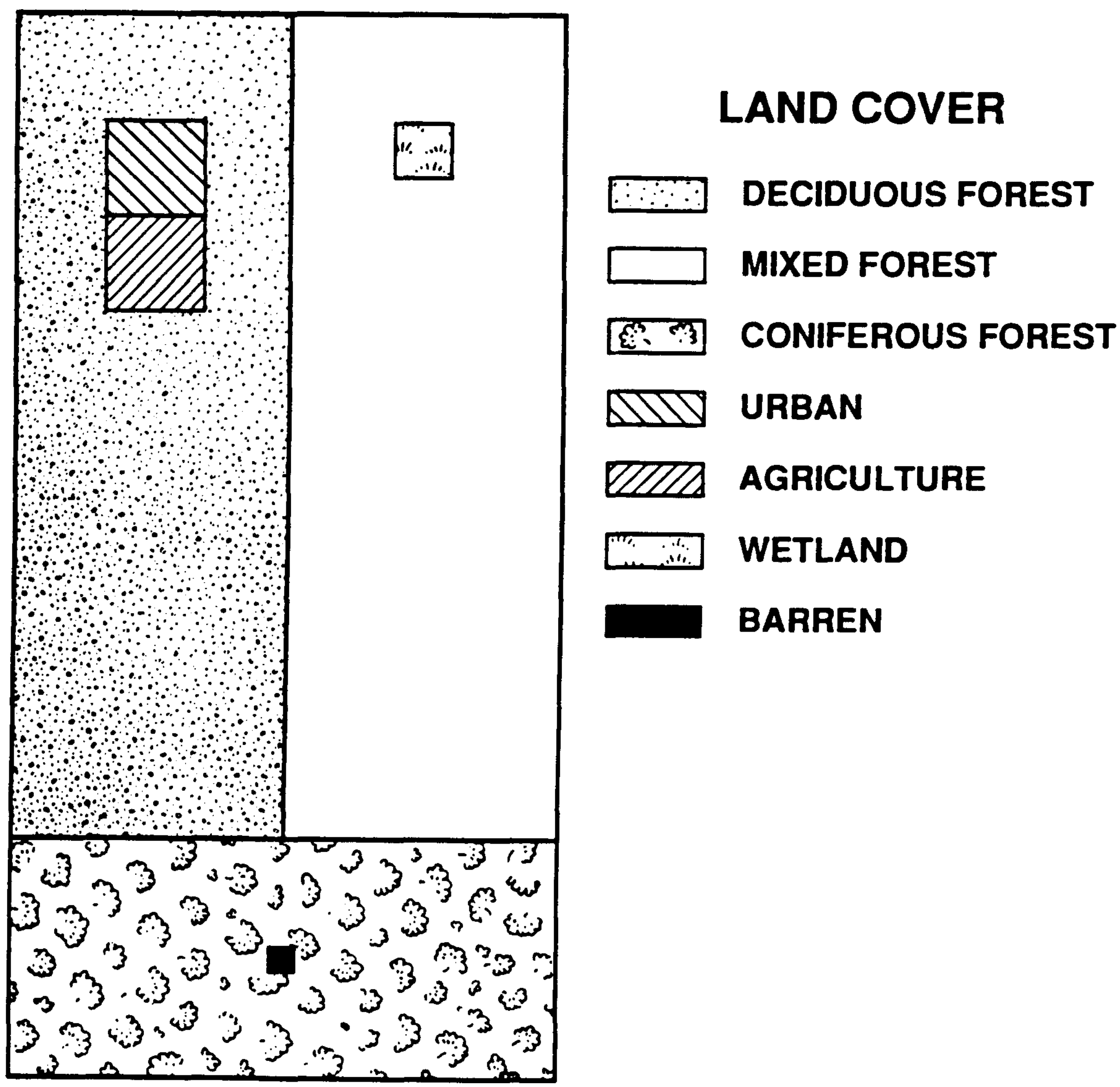

Fig. 3-3. Generalized representation of blocked baseline landscape.

3.3.2 Endpoints

Table 3-1 describes the 18 endpoints selected for this demonstration. Most of our endpoints are likely to be used as measurement endpoints rather than assessment endpoints (see Chapter 2). Measurement endpoints include the obvious changes in the percentage of selected land cover types. Edge habitat is sensitive to the dissection of the landscape created by bark beetle patches. Forest interior, defined as the sum of all forest cells surrounded on all four sides by forest, is also sensitive to dissection. The landscape indices, contagion and dominance, describe overall patterns on the landscape (O'Neill et al. 1988). High values of contagion indicate large contiguous patches. The landscape is dissected into many small patches when contagion is low. High values of dominance indicate a landscape dominated by one or two land covers. Land covers are found at approximately equal proportions when the value of dominance is low.

Two water quality endpoints are considered. The lake pH shift is a simple measurement endpoint that indicates a significant (greater than measurement error) change in pH. The acid improvement endpoint relates to aquatic resources and is thus an assessment endpoint. Most fish species do not reproduce in lakes with pH values below 5.5. Only six of the 66 lakes had initial pH values less than 5.5.

3.3.3 Exposure Assessment

For this demonstration, source terms were not developed since ambient ozone concentrations were the hazard of interest. Two ambient ozone exposure scenarios were used in the analysis. The high ozone scenario assumes a maximum 7-hour average ozone concentration of 0.090 ppm during the growing season. The 0.090-ppm value approximates the highest maximum 7-hour average ozone concentration experienced during a growing season in New York State [New York State Department of Environmental Conservation (NYSDEC) 1986] and therefore was selected as an upper bound for the seasonal average for the purposes of this analysis. The low ozone scenario assumes a maximum 7-hour average ozone concentration of 0.024 ppm during the growing season (NYSDEC 1986). Both scenarios assume uniform ozone exposure across the study region (all grid cells have the same value).

3.3.4 Effects Assessment

For a single iteration of the Monte Carlo model of ozone-induced bark beetle attacks, the probability of bark beetles attacking a coniferous or mixed conifer-hardwood grid cell was assumed to be 0.01 under the low ozone scenario and 0.03 under the high ozone scenario. The size distribution of patches (1 to 15 cells or 4 to 60 ha) under each scenario was based on studies of southern pine beetle infestation (Thatcher et al. 1980, Coster and Searcy 1980), while the frequency of patches (e.g., number of bark beetle attacks) was based on research in California relating bark-beetle-induced mortality in western conifers to ambient ozone concentrations (USEPA 1987, USEPA 1986b, Stark et al. 1968, Miller et al. 1969). Each attack was assumed to spread to kill all the conifer trees within a patch.

Under the low ozone scenario, an average of 4% of the land area (frequency x average patch size) occupied by coniferous forest or mixed conifer-hardwood forest was affected. Under the high ozone scenario, an average of 12% was affected.

For each Monte Carlo iteration of the spatial simulation model, bark beetle outbreak epicenters were randomly assigned to susceptible forest cells (conifers or mixed) with a probability appropriate to the ozone scenario (Figure 3-4a,b). Once a conifer or mixed conifer-hardwood cell was selected to be a beetle outbreak epicenter, the patch size associated with that epicenter was randomly determined by using the patch size distribution appropriate to that ozone scenario (Figure 3-4c). The model then converted the surrounding forest cells by moving in a clockwise direction around the epicenter until the appropriate patch size was created (Figure 3-4d). Coniferous forest cells were converted to shrubland cells, while mixed-conifer-hardwood cells were converted to deciduous forest cells. If the land cover pattern in the vicinity of the epicenter was such that a contiguous patch of the designated size could not be created, the model created the largest contiguous patch possible. Once the forest cover changes in the entire landscape had been modeled, the terrestrial endpoint values were calculated for that iteration (Figures 3-5).

The aquatic endpoint values for each iteration were calculated by linking the land cover output of the terrestrial landscape model to a lake water quality model developed for headwater lakes in the Adirondacks. The water quality regression model is sensitive to the amount and type of forest cover within a watershed (Hunsaker et al. 1986). This model

ORNL-DWG 88M-13649

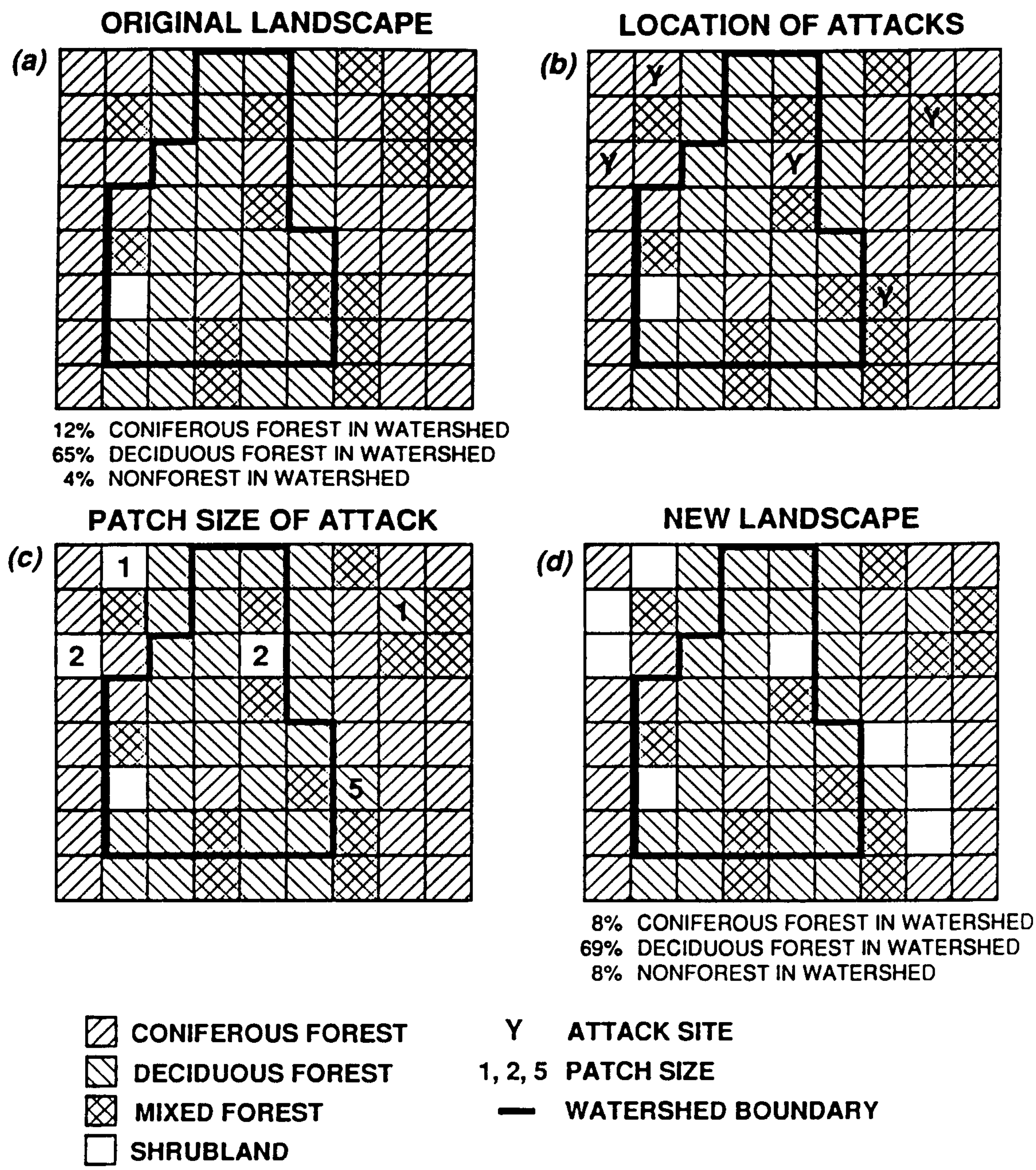

Fig. 3-4. Illustration of spatial model simulating ozone-induced land cover changes. Location of attacks and patch size are based on probability distributions that are specific to the ozone scenario. (a) Baseline landscape. (b) Randomly located attack sites. (c) Randomly assigned patch sizes where location of number indicates initial cell attacked and size of number indicates maximum number of cells that could change. (d) Landscape after beetle attack.

ORNL-DWG 88M-13650

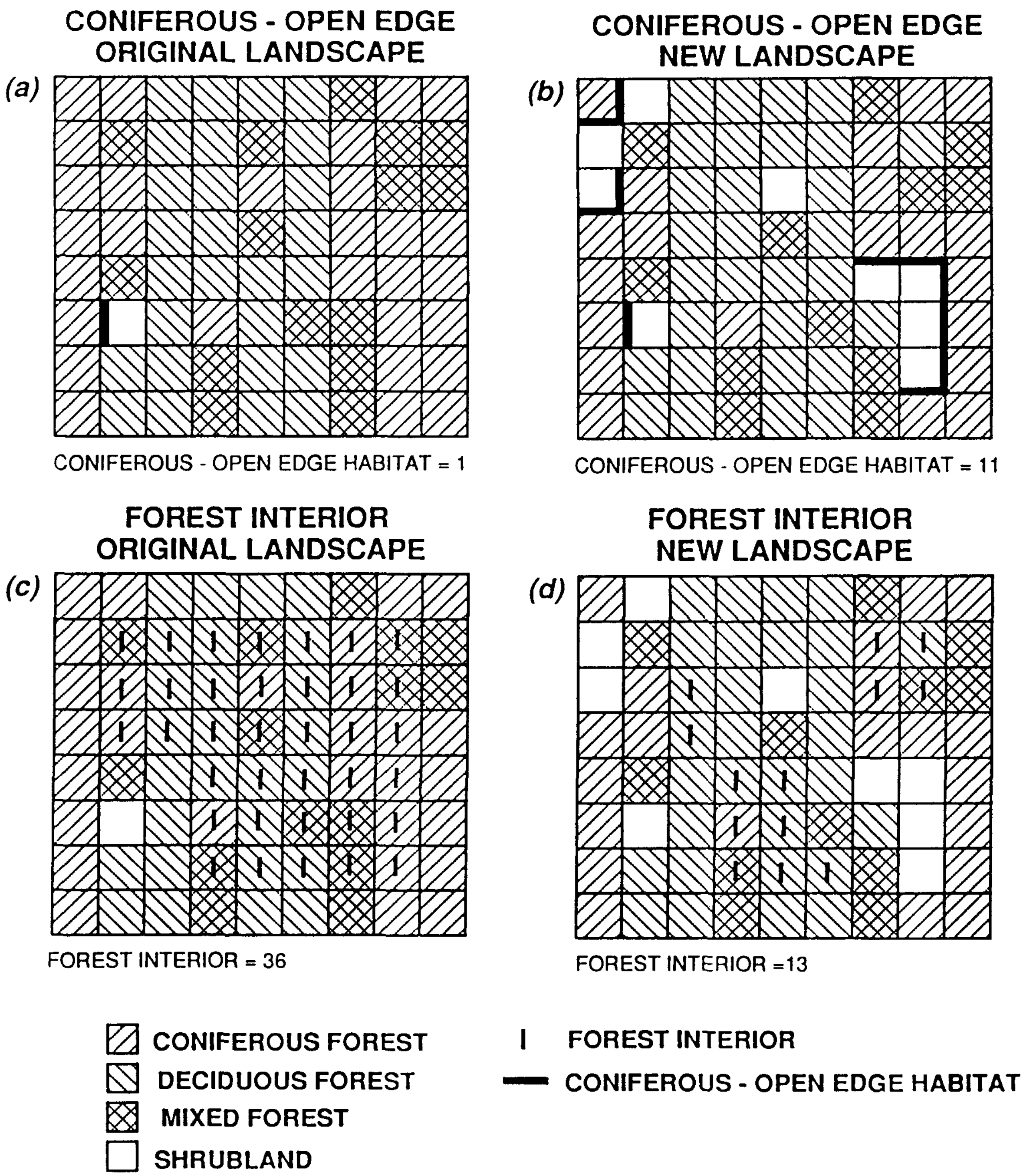

Fig. 3-5. Illustration of endpoint changes resulting from ozone-induced land cover changes. (a) Conifer-open edge in original landscape. (b) Conifer-open edge after beetle attack. (c) Interior forest in original landscape. (d) Interior forest after beetle attack.

requires 31 variables, only 3 of which were altered by the land cover disturbance model in this demonstration (percentages of conifer forest cover, deciduous forest cover, and nonforest cover). After each iteration of the spatial landscape model, these three variables were calculated for each lake watershed (Figure 3-4d). These values were then used in the lake water quality model to calculate the impact of the ozone disturbance on the pH of the lakes. The values for other variables in the regression model were taken from the Adirondacks Watershed Database maintained at Oak Ridge National Laboratory (Hunsaker et al. 1986). Values for the two water quality endpoints were then calculated from lake pH values (Table 3-1).

3.3.5 Risk Analysis

A value for each endpoint was calculated from the three baseline landscapes (see Table 3-3). Baseline values were compared with the endpoint values generated by the assessment model to determine the fraction of the Monte Carlo model iterations in which the endpoint measure changed by more than ±10% or ±25%. These fractions were then used to calculate the risk or probability of a low or high ozone scenario having a detectable or significant effect on a given endpoint (Tables 3-4 through 3-6). The detectable (±10%) and significant (±25%) values were arbitrarily selected for this demonstration.

Table 3-4. The risk or probability (in percent) of exceeding the endpoints and the values of endpoint measures for the actual landscape. For this assessment, risk is defined as the probability of a change greater than either 10% or 25% of the original value of the endpoint measure. The values of the endpoint measure for the baseline, low ozone, and high ozone scenarios are listed. Definitions of endpoints are given in Table 3-1.

	Ozone Scenarios				Value of Endpoint Measure		
	Low Scenario		High Scenario		Base-line	Low Ozone[a]	High Ozone[a]
Risk Criterion	≥10%	≥25%	≥10%	≥25%	(Actual)	(Mean)	(Mean)
COVER ENDPOINTS							
Forest	0	0	0	0	93.6	93.1	91.3
Deciduous	0	0	100	0	39.8	41.2	45.6
Coniferous	0	0	19	0	21.6	21.1	19.5
Mixed	0	0	36	0	38.6	37.8	34.9
EDGE HABITAT ENDPOINTS							
Deciduous-open	100	32	100	100	67	82	132
Coniferous-open	100	100	100	100	143	238	402
Mixed-open	0	0	23	0	152	156	164
Dec-agriculture	51	1	100	93	15	16	21
Con-agriculture	0	0	45	0	73	71	66
Mix-agriculture	0	0	52	0	57	56	51
Dec-wetlands	1	0	54	0	15	15	16
Con-wetlands	7	0	44	3	10	10	10
Mix-wetlands	1	0	44	0	15	15	14
FOREST INTERIOR	0	0	7	0	260,000	251,400	235,600
LANDSCAPE INDICES ENDPOINTS							
Dominance	0	0	0	0	1.47	1.59	1.51
Contagion	100	100	100	97	14.00	18.29	17.70
LAKE WATER QUALITY ENDPOINTS							
Lake pH shift[b]	1	0	89	0	0	4%	14%
Acid improvement[c]	67	28	97	89	0	18%	38%

[a]For 100 Monte Carlo iterations.
[b]Based on shifts for all 66 lakes.
[c]Based on shifts for the six lakes that had initial pH values ≤5.5.

Table 3-5. The risk or probability (in percent) of exceeding the endpoints and the values of endpoint measures for the random landscape. For this assessment, risk is defined as the probability of a change greater than either 10% or 25% of the original value of the endpoint measure. The values of the endpoint measure for the baseline, low ozone, and high ozone scenarios are listed.

	Ozone Scenarios				Value of Endpoint Measure		
	Low Scenario		High Scenario		Base-line	Low Ozone[a]	High Ozone[a]
Risk Criterion	≥10%	≥25%	≥10%	≥25%	(Random)	(Mean)	(Mean)
COVER ENDPOINTS							
Forest	0	0	0	0	92.2	91.5	89.6
Deciduous	0	0	100	0	36.3	37.8	42.2
Coniferous	0	0	2	0	24.1	23.6	22.0
Mixed	0	0	19	0	39.5	38.6	35.9
EDGE HABITAT ENDPOINTS							
Deciduous-open	100	0	100	100	163	191	277
Coniferous-open	100	100	100	100	216	302	443
Mixed-open	0	0	46	0	281	290	310
Dec-agriculture	2	0	100	9	53	56	63
Con-agriculture	0	0	58	0	89	86	80
Mix-agriculture	0	0	51	0	102	99	92
Dec-wetlands	0	0	93	2	45	47	52
Con-wetlands	0	0	48	0	36	35	32
Mix-wetlands	0	0	56	0	70	68	62
FOREST INTERIOR	0	0	99	0	241,800	232,800	215,700
LANDSCAPE INDICES ENDPOINTS							
Dominance	0	0	0	0	1.40	1.52	1.44
Contagion	100	100	100	94	13.62	17.79	17.17
LAKE WATER QUALITY ENDPOINTS							
Lake pH shift[b]	1	0	72	0	66	3%	12%
Acid improvement[c]	26	3	64	27	5	6%	19%

[a]For 100 Monte Carlo iterations.
[b]Based on shifts for all 66 lakes.
[c]Based on shifts for the five lakes that had initial pH values ≤5.5.

Table 3-6. The risk or probability of exceeding the endpoints and the values of endpoint measures for the blocked landscape. For this assessment, risk is defined as the probability of a change greater than either 10% or 25% of the original value of the endpoint measure. The values of the endpoint measure for the baseline, low ozone, and high ozone scenarios are listed.

	Ozone Scenarios				Value of Endpoint Measure		
	Low Scenario		High Scenario		Base-line	Low Ozone[a]	High Ozone[a]
Risk Criterion	≥10%	≥25%	≥10%	≥25%	(Blocked)	(Mean)	(Mean)
COVER ENDPOINTS							
Forest	0	0	0	0	93.6	93.0	91.1
Deciduous	0	0	100	0	38.0	39.4	43.9
Coniferous	0	0	24	0	21.9	21.4	19.8
Mixed	0	0	13	0	40.1	39.2	36.3
EDGE HABITAT ENDPOINTS							
Deciduous-open	1	0	66	0	23	23	125
Coniferous-open	100	100	100	100	4	121	345
Mixed-open	3	0	28	6	8	8	8
Dec-agriculture	0	0	0	0	16	16	16
Con-agriculture	0	0	0	0	0	0	0
Mix-agriculture	0	100	0	0	0	0	0
Dec-wetlands	100	0	86	86	0	<1	<1
Con-wetlands	0	0	0	0	0	0	0
Mix-wetlands	8	0	45	6	8	8	7
FOREST INTERIOR	0	0	0	0	273,300	264,290	248,631
LANDSCAPE INDICES ENDPOINTS							
Dominance	0	0	0	0	1.47	1.58	1.51
Contagion	100	3	100	0	18.62	23.14	22.73
LAKE WATER QUALITY ENDPOINTS							
Lake pH shift[b]	61	0	100	13	66	11%	21%
Acid improvement[c]	100	72	100	100	10	32%	58%

[a]For 100 Monte Carlo iterations.
[b]Based on shifts for all 66 lakes.
[c]Based on shifts for the ten lakes that had initial pH values ≤5.5.

3.4 RESULTS

3.4.1 Actual Landscape

As expected, high ozone led to a significant risk of a ≥10% decrease in conifers and a consequent increase in deciduous trees as mixed stands were converted to deciduous stands (Table 3-4). Under the stress imposed in this study, there is a negligible risk of a ≥25% change in forest composition. Overall forest cover was insensitive to ozone.

The amount of forest edge habitat was sensitive to both ozone scenarios, especially the conifer edge habitat. Forest interior habitat was not sensitive. The ozone scenarios did not affect the dominance of land cover types in the region but did substantially affect landscape pattern by increasing landscape contagion.

The ozone-induced changes in forest cover had significant effects on lake water quality, especially on the limited number of lakes that had initial pH values less than or equal to 5.5. In 89 of the 100 model iterations (probability of 89%) under the high ozone scenario, >10% of the lakes experienced an increase in pH of greater than 0.2 pH units. In no instance did 25% or more of the lakes shift in response to ozone. The six lakes that had initial pH values less than or equal to 5.5 were quite sensitive to the forest composition shifts in response to ozone. Under the low ozone scenario, more than 10% of the lakes were raised above a pH of 5.5 67% of the time (e.g., 67 times out of 100), while more than 25% of the lakes were raised above a pH of 5.5 28% of the time. With the high ozone scenario, on the average, half of the low pH lakes were raised above a pH of 5.5.

3.4.2 Altered Landscapes

The sensitivity of disturbance effects to initial landscape pattern was evaluated by imposing the same disturbance scenarios onto the three baseline landscapes--actual, random, and blocked. The risk or probability of exceeding the endpoints was then compared for these landscapes (Tables 3-7 and 3-8). The baseline landscapes contain essentially the same percentage of each land cover, and thus dominance does not vary significantly. These landscapes represent points on a continuum from random to actual to blocked. The random landscape is very fragmented with a low contagion value, low amount of forest interior, and high amount of edge, whereas the blocked landscape has a high value of contagion, high amount of interior, and low amount of edge (Table 3-3).

The random and actual landscape endpoints have a similar pattern of risk for the disturbance scenarios although the edges and forest interior are at greater risk for the random landscape. The risk for endpoints in the blocked landscape is very different; one should keep in mind that the direction of the risk is dependent on the way the initial blocked pattern was set up. In the blocked landscape, contagion and forest interior are at lower risk to disturbance; when baseline values are high, these endpoints are less susceptible. Lake endpoints are dependent on watershed locations, and the more heterogeneous the watershed landscape the less susceptible any one lake is to the disturbance. Thus lakes in the random landscape are the least likely to change. In the blocked landscape, all lakes with total watersheds of coniferous or mixed forest that are hit by the disturbance will probably undergo a significant change.

Table 3-7. Influence of baseline landscape pattern on risk to endpoints when using the low ozone scenario and a probability of change greater than or equal to 10%

Endpoints	Actual	Random	Blocked
EDGE HABITAT			
Deciduous-open	100	100	1
Coniferous-open	100	100	100
Mixed-open	0	0	3
Deciduous-agriculture	51	0	0
Coniferous-agriculture	0	0	0
Mixed-agriculture	0	0	0
Deciduous-wetlands	1	0	100
Coniferous-wetlands	7	0	0
Mixed-wetlands	1	0	8
FOREST INTERIOR	0	0	0
LANDSCAPE INDICES			
Dominance	0	0	0
Contagion	100	100	100
LAKE WATER QUALITY ENDPOINTS			
Lake pH shift	1	1	61
Acid improvement	67	26	100

Table 3-8. Influence of baseline landscape pattern on risk to endpoints when using the high ozone scenario and a probability of change greater than or equal to 10%

Endpoints	Actual	Random	Blocked
EDGE HABITAT			
Deciduous-open	100	100	66
Coniferous-open	100	100	100
Mixed-open	23	46	28
Deciduous-agriculture	100	100	0
Coniferous-agriculture	45	58	0
Mixed-agriculture	52	51	0
Deciduous-wetlands	54	93	86
Coniferous-wetlands	44	48	0
Mixed-wetlands	44	56	45
FOREST INTERIOR	7	99	0
LANDSCAPE INDICES			
Dominance	0	0	0
Contagion	100	100	100
LAKE WATER QUALITY ENDPOINTS			
Lake pH shift	89	72	100
Acid improvement	97	64	100

3.5 DISCUSSION

Pattern-related endpoints (edge habitat, interior forest habitat, and contagion) used in this study are examples of measurement endpoints (see Chapter 2). They can be related to assessment endpoints such as deer abundance and bird populations. For example, our research showed that the abundances of several bird species were significantly related to coniferous forest edge in three physiographic provinces of Georgia. As edge increased by a factor of 2, the abundance of these species increased by a factor of 2 to 10.

We also know from this work that some species can be more sensitive to the landscape pattern than to the land cover. For example, four of the five bird species related to coniferous-forest-edge habitat were not related to total coniferous forest cover. Of the 43 Georgia bird species examined, more species were related to pattern than to landscape cover attributes. That is, the manner in which a resource such as coniferous forest was arrayed in the landscape had more influence on bird abundance than did the total amount of the resource.

Landscape pattern has been shown to affect a variety of potential ecological endpoints. Loss of merchantable timber due to blowdown in the Pacific Northwest is linked to harvest cutting patterns (Franklin and Forman 1987). Shrimp harvest offshore the Mississippi River delta is more related to the amount of wetland edge in the delta than the total amount of wetland (Browder et al. 1988). Woodland bird species abundances in two regions of the Netherlands are related to not only the amount of woodlands but also the size and spatial distributions of the woodlots (van Dorp and Opdam 1987). The regional abundance of many wildlife species that prefer forest interior habitat is related to the abundance of interior forest (Rosenberg and Raphael 1984).

The results of this risk analysis support our contention that the consideration of landscape pattern is necessary for regional ecological risk assessment (Hunsaker et al. 1988). In this analysis, endpoints that were dependent on landscape pattern were at greater risk than endpoints that were independent of landscape pattern. Some endpoints are more sensitive to the initial landscape pattern. An analysis which ignored spatial pattern would have concluded that there was little or no risk associated with the ozone hazard, whereas our analysis showed that the hazard significantly altered important resource habitat in this region.

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